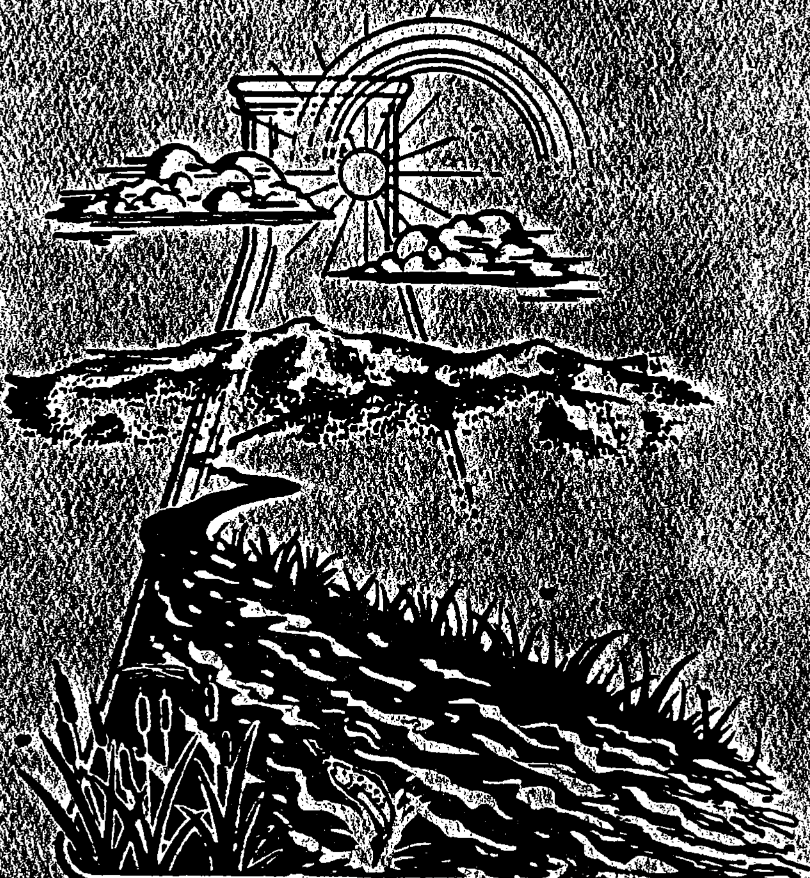
MANUAL FOR IMPLEMENTATION AND DEVELOPMENT OF AQUATIC RESOURCE INVENTORY AND MONITORING METHODOLOGY IN PRAIRIE PARKS
WILSON'S CREEK NATIONAL BATTLEFIELD



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INVENTORY AND MONITORING

Management & Technical: Responsibilities & Partnership

Recent interest within the National Park Service for a more comprehensive and systematic inventory of natural resources and periodic monitoring to determine condition and to assess any man-induced changes has been manifested in the Natural Resources Assessment Action Program. Many parks have some resource inventory and monitoring (I&M) activities; however, many of these same parks are often at a loss to interpret collected environmental data when confronted with some of the more pervasive, complex threats to the integrity of their resources. In the past, many inventories of natural resources have been little more than lists of species and locations of samples sites; many monitoring efforts have been restricted to those mandated by human health concerns comprising a loose collection of assorted chemical constituents, often directed at concerns about water quality.

This article discusses the integration of management and technical aspects, and introduces a framework for conducting resource inventories and subsequent monitoring. Inventory and monitoring in the NPS is a responsibility shared between resource management and research. Therefore, any program conducting environmental inventory and subsequent monitoring efforts should have certain critical elements that include: 1) a procedure for establishing clear co-responsibilities for resource managers and research scientists, 2) a concept of the ecological paradigm as a framework for the collection of information and interpretation of data in a resource inventory program, and 3) a rationale for using data generated in monitoring programs.

A primary role of resource managers is to secure information on threats and identify the needs for managing natural resources. Resource management needs should be put into a clear set of management objectives. It is the role of the research scientist to translate these management objectives into scientific hypotheses, data collection procedures, and conclusions. These two sets of activities should be considered parallel and interacting.

I. The initial step is a clear statement of resource management programmatic concerns by the Park. These concerns may be mandated by legislation establishing the Park, come from pervasive threats such as pollution, result from land-use changes outside the Park boundaries, or be the consequences of management decisions. It is imperative that the statement of natural resource management concerns be both explicit and inclusive because it leads directly to and drives the other steps in the I&M process. This statement of concerns should be a joint effort between resource management and a research scientist. These individual resource management concerns should be arranged by priority to assign available resources.

II. The second step is the formulation of scientific tasks that are directly predicated from the resource management concerns in the first step. The set of scientific tasks should address the prioritized Park natural resources concerns, and in certain cases may also address broader issues that include long-term issues of regional or national importance. An explicit framework to implement these scientific tasks is as follows:

A) The scientific tasks should be in the form of explicit sets of testable hypotheses that address the Park's specific and broader NPS resource management concerns.

B) These hypotheses will entail specific data set requirements. The level of effort in the data collection will be directly related to how the data will be used, i.e. for general knowledge and interpretation vs. for legal defense of resources.

C) At this point there will be a comparison of resources available and costs against the prioritized list of natural resource concerns to be addressed. In effect, this step will determine what tasks can be done. Both resource managers and scientific personnel should reach a consensus on the content of the final scope of work.

D) The data collection should have a Quality Assurance and Quality Control program in place before sampling takes place. This includes a precise statement of procedures of how data are to be collected, instrumentation used, use of chemicals or biological standards, and limits of detection.

E) There should be explicit plans for data storage, preservation, and archiving of appropriate samples. Considerations should be addressed as to who will be using the data and the time frame in which the data will be used.

F) The data should be subject to quantitative analysis that both establishes a basis for statistical parameters and forms a basis for accepting or rejecting the hypotheses generated by the resource management concerns.

G) Consideration of the final products in terms of written reports should be given before the inventory is begun. These products may range from a report written directly to the Park or the Regional Science Office, articles submitted to the peer reviewed literature, extensive materials covered in a monograph, or legal depositions and expert testimony.

This brief article cannot begin to cover inclusively all the technical aspects of inventories, however, this data should represent more than tabulations of data points in time and space. The NPS Resource Inventory and Monitoring Program must be designed to integrate the management concerns with a scientific/technical effort that identifies and quantifies key components of those resources with some level of ecological understanding that will allow both scientific comprehension and management interpretation. Here, we would like to introduce the term **ecological paradigm**, defined as a specific model or set of hypotheses encompassing comprehensive characterization of resources at the population, community, or ecosystem level of organization. There must be an ecological basis to the inventories that can explain changes in measured attributes of natural resources due to natural variability, naturally induced change, and man-induced change.

Environmental monitoring of important natural resources is an extension of resource inventories. The same partnership and general procedures integrating management and scientific personnel in inventories should be developed to pose questions for monitoring programs. Resource managers are ultimately responsible for the questions posed by a monitoring program and final questions as to how the data will be used. Monitoring, by definition, is limited in magnitude and frequency of data collected. Resource management should be especially aware of the limits and conflicts inherent in any monitoring program. Two general opposing strategies of monitoring are 1) distinguishing departures from standard conditions, detections of violations of standards, or 2) establishing changes in conditions or key variables over time. These two contrasting activities have conflicting data requirements. Where resources are limited the environmental questions should be elaborated before the monitoring program begins.

This outline of technical aspects is not intended to be exhaustive of all possible aspects of natural resource inventories or environmental monitoring, nor should a resource inventory in a Park necessarily include all of the items elaborated in the lists. Inventory and monitoring in the National Park System is a relatively new concept and practice, and one that will vary among parks and evolve with time. These are essential steps that assign specific roles to resource managers and scientists to help insure useful knowledge in the managing of Park resources.

Inventory & Monitoring in a Risk Analysis Framework

Risk analysis in the life sciences has focused on the factors directly affecting human health and mortality. For example, calculations can be made on the risk to the human population due to factors such as smoking, the use of certain medicines, or the development of nuclear power generating plants. In the field of environmental sciences, risk analysis has been applied to the registration and use of pesticides, the release of industrial chemicals using estimations of chemical fate, and batteries of laboratory toxicity tests to estimate the direct effects of such chemicals on important resource organisms. These methods of assessment do not, however, estimate indirect effects on other organisms, effects at higher levels of organization such as community or ecosystem nor do they estimate the effects from non-chemical and multiple impacts.

Natural resource inventories in the past have been efforts to make catalogs of existing species, determine their ecological distribution, the limits of their geographical ranges, and stock assessments of major resource species, such as volume of timber or numbers of fish. Natural resource inventories provide key elements in an ecological risk analysis program that will provide risk assessment if several steps are included.

The strategy for assessing and managing risks involving ecological change using natural resource inventories includes:

1] Development of the concept of **ecosystem health,**

2] Develop **regional profiles** of the health of ecosystems in terms of critical ecological characteristics and,

3] Formulation of ecological measurement **endpoints** to determine ecological health and provide a basis for monitoring.

The concept of *ecosystem health* has been suggested by several authors as a basis for environmental assessment and more recently for consideration in the NPS inventory and monitoring program. Operationally the concept of ecosystem health should include determination of keystone species, critical communities, and important ecosystem-level processes, and **not** an elaboration and evaluation of overwhelming numbers of species. This will require some application of critical judgement and prioritization by ecologists conducting inventories. Ecological risk analysis is based on two assumptions 1) Threats to NPS natural resources are unavoidable due to development, and 2) Management of natural resources is, in practice, based on a paucity of scientific information.

In defining ecological health, an analogy to characterizing human health is useful. Human health may be defined as the condition of the body as determined by a number of parameters such as certain physical characteristics as body temperature and structural integrity, height-weight relationship, the chemical condition of blood and other body fluids that are parameters of the biological condition of various organ systems etc. Measurement of these critical health attributes within certain ranges are considered healthy and when these limits are exceeded the individual is considered diseased or unhealthy, and some sort of medical action is warranted.

The ecological risk approach would use the information from resource inventories to identify resources and assess impact-induced changes before large scale damage has occurred. This form of assessment is retroactive in that it evaluates change that has already taken place, but is also proactive in that it uses estimations of change at an early stage and ecological knowledge from inventories to estimate the final state of a resource or ecosystem due to impacts from development. Condition of the resource or ecosystem rather than the source of stress is the beginning point of analysis.

Regional profiles of ecological health can be derived from regional biological data bases supplemented by park-specific inventory studies. Ecological theory provides the basis for management, diagnosis and treatment of the ecosystem in the same fashion that human anatomy and physiology provide the basis for the practice of human medicine. For example, at the population level single species routinely inventoried include major resource organisms such as commercially important fisheries or trees valued for timber, pest species, important charismatic megafauna, or endangered species. Inventories at the community level are commonly done to classify forest types, vegetation analysis, baseline data to establish biogeographic regions, and in assessment of environmental impact. Examples of paradigms at the ecosystem level are the River Continuum Concept which describes how physical, chemical, and biological conditions change as small streams become larger and larger eventually becoming a large river. Inventories of lake ecosystems can be facilitated by considering the trophic succession model (oligotrophic, mesotrophic, eutrophic, or hypertrophic), limiting nutrient, nutrient loading chemical classification, and by key physical attributes such as depth, volume, surface area, dynamics of mixing, formation and depth of thermocline.

In a fashion similar to human medicine approaches sets of *ecological endpoints* should be formulated from the ecological theory and data from regional profiles and Park inventories. An ecological endpoint is defined as an ecological parameter whose normal operating limits can be determined for the ecosystem in question. A major effort in ecosystem risk analysis is the identification of appropriate ecological endpoints that are indicators of ecological health and sensitive to early stages of anthropomorphic change. In order to relate to the values of human society these ecological endpoints should be associated with social, cultural, or economic consequences. From an operational aspect, ecological endpoints should be easily measured, quantifiable, and amenable to statistical analysis. They should be sensitive to early or defined status of ecological stress to allow for mitigation measures to be employed.

Examples of endpoints include quantitative estimates of density of commercially important species or charismatic megafauna, invasion or introduction of pest species on a large scale, changes in the concentration of critical nutrients (either additions or losses), especially those nutrients such as nitrogen and phosphorus known to control primary production, and measures of community structure such as density, species richness, diversity, similarity, and dominance. Ecological endpoints can be arranged in diagnostic profiles to characterize and monitor through time the ecological health of the resource and ecosystem in question. Some research in determining appropriate ecological endpoints, statistical properties, and required measurement frequency are necessary.

Biological Criteria for Endpoints

A principal objective of the Clean Water Act (CWA) is to restore and maintain the physical, chemical, and biological integrity of surface waters. Although this goal is fundamentally biological in nature the specific methods to reach this goal have been predominated by such non-biological measures as chemical/physical water quality. The rationale for this process is well known - chemical criteria developed through toxicological studies of representative aquatic organisms serve as surrogates for measuring the attainment of the biological goals of the CWA. The presumption is that improvements in chemical water quality will be followed by a restoration of biological integrity. Although this type of approach may give the impression of empirical validity and legal defensibility it does not directly measure the ecological health and well-being of surface waters. Recent information shows that other factors in addition to chemical water quality are responsible for the continuing decline of surface water resources. Because biological integrity is affected by these factors in addition to chemical water quality, controlling chemical discharges alone does not in itself assure the restoration of biological integrity.

This draft addresses the use of community level biological criteria as endpoints in the assessment and protection of aquatic life. Community level endpoints cannot perform every task necessary in a water quality assessment program, however, they do offer some significant advantages over the traditional chemical and/or bioassay approaches alone. The addition of biological criteria can be a valuable aid in supporting other assessment methods if they are combined in a truly integrated program. It is important to recognize and exploit the links between the chemical, bioassay, and community level data collected as part of a biosurvey approach to water quality assessment. Direct quantitative assessment of biological communities significantly broadens the base from which the NPS can manage and protect surface water resources. This approach is compatible with the biological goals of the Water Quality Act and the important role that biological principles have in water resource management in general.

The Rationale for Community Level Endpoints

The existing condition of the biota resident in any surface water body is the integrated result of many chemical, physical, and biological processes over time. Thus the existing biological condition is the cumulative a result of these processes. Biological communities are indicators of environmental conditions since they inhabit the receiving waters continuously and are subject to natural and anthropogenic chemical and physical influences that occur over time. The community level assessment approach represents an evaluation strategy that is used to characterize the critical ecological components of the chemical, physical, and biological processes that affect biological performance.

Defining Biological Integrity

Biological integrity is considered relative to 1) conditions that existed prior to human civilization, 2) the protection and propagation of balanced, indigenous populations and communities, and 3) ecosystems that are unperturbed by human activities. This criteria (at least 1 and 3) refer to a pristine condition that probably exists in few, if any, ecosystems in the conterminous United States. One U.S. EPA sponsored work group concluded that biological integrity, when defined as some pristine condition, is difficult if not impractical to precisely define and assess. The pristine definition of biological integrity is considered a conceptual goal toward which pollution abatement efforts should strive, although current, past, and future uses of surface waters may prevent its full realization. Biological integrity can be scientifically characterized as the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region. This is a workable definition of biological integrity that is based on measurable characteristics of biological community structure and function in least impacted habitats. It also provides the fundamental underlying theory for the eventual development of community level endpoints using the biosurvey/ecoregion approach. Systems that possess or reflect biological integrity can withstand or rapidly recover from most perturbations imposed by natural environmental processes and some of those induced by humans. The reaction of an aquatic ecosystem to stress depends largely on the frequency, magnitude, and duration of the effect and the inherent sensitivity of the system itself. Biological communities that are degraded and therefore lack integrity have had their capacity to withstand and rapidly recover from stress exceeded. Some communities are likely to become even further degraded under incremental increases in stress. In contrast communities that reflect biological integrity do so because their capacity to withstand stress has not been exceeded to result in a temporally extended degradation of structural or functional organization. A biological system can be considered to have integrity when its inherent potential is realized, its condition is stable, its capacity for self-repair when stressed is preserved, and minimal external support for management is needed.

Maintenance of Biological Integrity

Alterations of the physical, chemical, or biological processes supporting aquatic ecosystems may adversely affect aquatic biota and therefore the biological integrity of the water body. Efforts to protect and restore water resources that focus on only one or two critical components will fail if other factors are wholly or partially responsible for the observed degradation. Efforts to maintain and improve the quality of surface water resources in general and aquatic life in particular must be guided by methods and monitoring that identify stress on a multitude of factors that together support aquatic natural resources.

Applicability of Biological Criteria

Some communities, particularly fish and many macroinvertebrates that inhabit the receiving waters continuously, are a reflection of the chemical, physical, and biological history of the receiving waters. Many fish species and invertebrate taxa have life spans of several years (2-10 yrs. and longer), thus the condition of the biota is an indication of past and recent environmental conditions. Biological surveys need not be conducted under absolute "worst case" conditions to provide a comprehensive and meaningful evaluation. A finding that biological integrity is being achieved not only reflects the current healthy condition, but also means that the community has withstood and recovered from any short-term stresses that may have occurred prior to field sampling. Biological assessment techniques have progressed to the point that incremental degrees and types of degradation can be determined and presented as numerical evaluations (e.g. Index of Biotic Integrity-IBI) that have relative meaning to non-biologists. Chemical criteria and bioassay application techniques will always play an important role in water quality regulation. Their value, however, is greatly enhanced when used in combination with holistic assessments of the resident biota. Biological communities are broader indicators of environmental problems than is chemical sampling alone because they reflect the integrated dynamics of chemical, physical, and biological processes.

Biosurveys, Bioassays, & Chemical Monitoring

The water quality-based approach to pollution assessment requires various types of data. Community level assessments are best used for detecting aquatic life impairments and assessing their relative severity. Once an impairment is detected additional chemical and biological (toxicity) testing is necessary to identify the causative agent and its source, and to implement appropriate mitigation. Following mitigation, biosurveys are important for evaluating the effectiveness of such control measures. Some of the advantages of using biosurveys for this type of monitoring are:

1. Biological communities reflect overall ecological integrity (i.e., chemical and physical, as well as biological integrity). Therefore, biosurvey results directly assess the status of a waterbody relative to the primary goal of the Clean Water Act.

2. Biological communities integrate the effects of different pollutant stressors and thus provide a holistic measure of their aggregate impact. Communities also integrate the stresses over time and provide an ecological measure of fluctuating environmental conditions. The integrated response of biological communities to highly variable pollutant inputs offers a particularly useful method for monitoring nonpoint-source impacts and the effectiveness of certain Best Management Practices.

3. Routine monitoring of biological communities can be relatively inexpensive, particularly when compared to the cost of assessing toxic pollutants, either chemically or with toxicity tests.

4. The status of biological communities is of direct interest to the public as a measure of a pollution free environment, while reductions in chemical pollutant loadings are not as readily understood by the layman as positive environmental results.

5. Where methods for assessing specific ambient impacts do not exist, for example nonpoint-source and impacts that degrade habitat, assessing biological communities may be the only rational means of evaluation.

Biosurvey methods have a longstanding history of use for before and after and upstream-downstream monitoring. However, the intermediate steps in pollution control, identifying causes and limiting sources, require information of a more complex strategy of chemical, physical, and/or additional biological data. These data are needed to identify the specific stress agents causing impact. This may be a relatively simple task, however, given the array of potentially important pollutants and their possible combinations, it may to be both difficult and costly. In situations where specific chemical stress agents are either poorly understood or too varied to assess individually, toxicity tests can be used to focus specific chemical investigations or to characterize generic stress agents, for example at Wilson's Creek National Battlefield (see paper in additional readings). Although biosurveys can be used to help locate the likely origins of impact, chemical analyses and/or toxicity tests are usually necessary to confirm the responsible sources and develop appropriate discharge limits.

Effective implementation of the water quality-based approach requires that various monitoring techniques be considered within a larger context of water resource management. Both biological and chemical methods play critical roles in a successful pollution control program. They should be considered complementary rather than mutually exclusive approaches that will enhance overall program effectiveness when used appropriately.

Different Communities in Biosurveys

The bioassessment techniques presented in this document focus on the evaluation of water quality, habitat, and benthic macroinvertebrate and fish community parameters. Many state water quality agencies employ trained and experienced benthic biologists, have accumulated considerable background data on macroinvertebrates, and consider benthic surveys a useful assessment tool. However, water quality standards, legislative mandates, and public opinion are more directly related to the status of a waterbody as a fishery resource. The integration of functional and structural/compositional metrics, which forms the basis for the IBI is a common element to the fish and benthic rapid bioassessment approaches.

Although no methods are presented here for conducting algal assessments, algal communities are also useful for water quality monitoring. They represent another trophic level, exhibit a different range of sensitivities, and will often indicate effects only indirectly observed in the benthic and fish communities. As in the benthic macroinvertebrate and fish communities, integration of structural/compositional and functional characteristics provides the best means of assessing impairment.

Algal community structural/compositional analyses may be taxonomic or non-taxonomic. Taxonomic analyses (e.g., diversity indices, taxa richness, indicator species) are commonly used. Non-taxonomic measures, such as biomass and chlorophyll, can also be useful for detecting effects not indicated by taxonomic analysis. For example, toxic pollutants may cause sublethal (i.e., reproductive) effects which would not immediately be detected by taxonomic analyses such as taxa richness, but would be indicated by low biomass. In determining the taxonomic group or groups appropriate for a particular biomonitoring situation, the advantages of using each taxonomic group must be considered along with the objectives of the program. Some of the advantages of using macroinvertebrates and fish are presented.

__Advantages of Using Benthic Macroinvertebrates__

1. Macroinvertebrate communities are good indicators of localized conditions. Because many benthic macroinvertebrates have limited migration patterns or a sessile mode of life, they are particularly well suited for assessing site-specific impacts (upstream-downstream studies).

2. Macroinvertebrate communities integrate the effects of short-term environmental variations. Most species have a complex life cycle of approximately 1 year or more. Sensitive life stages will respond quickly to stress; the overall community will respond more slowly.

3. Degraded conditions can often be detected by an experienced biologist with only a cursory

examination of the macroinvertebrate community. Macroinvertebrates are relatively easy to identify to family; many intolerant taxa can be identified to lower taxonomic levels with ease.

4. Sampling is relatively easy, requires few people and inexpensive gear, and has no detrimental effect on the resident biota.

5. Benthic macroinvertebrates serve as a primary food source for many recreational and commercially important fish.

6. Benthic macroinvertebrates are abundant in most streams. Many small streams (lst and 2nd order), which naturally support a diverse macroinvertebrate fauna, only support a limited fish fauna.

7. Most State water quality agencies that routinely collect biosurvey data focus on macroinvertebrates. Many State water quality agencies have more expertise in aquatic entomology than in ichthyology.

Advantages of Using Fish

1. Fish are good indicators of long-term (several years) effects and broad habitat conditions because they are relatively long-lived and mobile.

2. Fish communities generally include a range of species that represent a variety of trophic levels (omnivores, herbivores, insectivores, planktivores, piscivores). They tend to integrate effects of lower trophic levels; thus, fish community structure is reflective of integrated environmental health.

Time for Benthic Collections

Monitoring is based on evaluation of relatively few samples at a site. Seasonality is particularly important when only a few collection sites are involved. The intent of a monitoring bioassessment is to evaluate overall biological condition, optimizing the use of the benthic community's capacity to reflect integrated environmental effects over time. Ideally, the optimal biological sampling season will correspond to reproductive cycles of the invertebrates. Maximum information for a benthic community is obtained when most benthic macroinvertebrates are within a size range (later instars) retained during standard sieving and sorting, and can be identified with the most confidence.

Reproductive periods and different life stages of aquatic insects are related to the abundance of particular food supplies. Peak emergence and reproduction typically occur in the spring and fall, although onset and duration vary somewhat across the United States. During peak reproduction, approximately 80 percent of the macroinvertebrates will be too small to be captured in sufficient numbers to accurately characterize the community. Additionally, food source requirements for early instars are different from those for later instars. Therefore, the biologically optimal sampling season would occur when the habitat is utilized most heavily by later instars and the food resource has stabilized to support a balanced indigenous community.

Natural and Artificial Substrates

The benthic assessment procedures employ direct sampling of natural substrates. However, where conditions are inappropriate for the collection of natural substrate samples, artificial substrates may be an option. Artificial substrates may be useful in situations such as large rivers, where an impact is attributable to physical alteration and channelization or chemical effects. Artificial substrates may be used to separate the two impact sources. Advantages and disadvantages of artificial substrates relative to the use of natural substrates are presented below.

Advantages of Sampling With Artificial Substrates

1. Artificial substrates allow sample collection in locations that are typically difficult to sample effectively (e.g., bedrock, boulder, or shifting substrates; deep or high velocity water).

2. As a passive sample collection device, artificial substrates permit standardized sampling by eliminating subjectivity in sample collection technique. Direct sampling of natural substrate requires similar effort and degree of efficiency for the collection of each sample. Use of artificial substrates requires standardization of setting and retrieval; however, colonization provides the actual sampling mechanism.
3. Confounding effects of habitat differences are minimized by providing a standardized microhabitat. Microhabitat standardization may promote selectivity for specific organisms if the artificial substrate provides a different microhabitat than that naturally available at a site. Most artificial substrates, by design, select for the Scraper and Filtering Collector communities. However, in some situations, accumulation of debris may cause a predominance of Collector-Gatherers.
4. Sampling variability is decreased due to a reduction in microhabitat patchiness, improving the potential for spatial and temporal similarity among samples.
5. Sample collection using artificial substrates may require less skill and training than direct sampling of natural substrates. Depending on the type of artificial substrate used, properly trained technicians could place and retrieve the substrates. However, an experienced specialist should be responsible for the selection of habitats and sample sites.

Disadvantages of Sampling With Artificial Substrates

1. Two trips (one to set and one to retrieve) are required for each artificial substrate sample; only one trip is necessary for direct sampling of the natural substrate. Artificial substrates require a long (8-week average) exposure period for colonization. This decreases their utility for certain rapid biological assessments.

2. Samples may not be fully representative of the benthic community at a station if the artificial substrate offers different microhabitats than those available in the natural substrate. Artificial substrates often selectively sample certain taxa, misrepresenting relative abundances of these taxa in the natural substrate. Artificial substrate samples would thus indicate colonization potential rather than the resident community structure. This could be advantageous if a study is designed to isolate water quality effects from substrate and other microhabitat effects. Where habitat quality is a limiting factor, artificial substrates could be used to discriminate between physical and chemical effects and assess a site's potential to support aquatic life on the basis of water quality alone.

3. Sample loss or perturbation commonly occurs due to sedimentation, extremely high or low flows, or vandalism during the relatively long (at least several weeks) exposure period required for colonization. Depending on the configuration of the artificial substrate used, transport and storage can be difficult. The number of artificial substrate samplers required for sample collection increases such inconvenience.

Quality Assurance/Quality Control

Effective quality assurance and quality control (QA/QC) procedures and a clear delineation of QA/QC responsibilities are essential to ensure the utility of environmental monitoring data. The term quality control refers to the routine application of procedures for obtaining prescribed standards of performance in the monitoring and measurement process. The term "quality assurance" includes the quality control functions and involves a totally integrated program for ensuring the reliability of monitoring and measurement data.

Statement of Work

The following contract or statement of work is an example that may be used to obtain scientific analysis of quantitative samples of macroinvertebrate communities which insures that the prescribed standards of performance in collection, sample processing, and monitoring are maintained.

Macroinvertebrate stream sampling analysis

Purpose: This contract is for the purpose of obtaining a scientific analysis of quantitative samples containing macroinvertebrates taken from streams in National Park Service properties. Information gained from this work will be used for longterm ecological monitoring and to build an aquatic biota species inventory.

Objectives: To provide a rapid turn-around of collected samples to help determine project conditions. In order to compare current samples with historic data, it is imperative that taxonomic nomenclature be consistent with park efforts.

Specifications: The contractor shall furnish all necessary equipment, personnel, and supplies required for the identification of benthic macroinvertebrates and water chemistry samples. To minimize inconsistencies between data generated through this contract and data previously generated by park personnel, references should be considered either as authoritative or as a guidebook.

Sample analysis

1) Number of samples: 5 quantitative macroinvertebrate samples per site (known as replicates) representing specified sampler types (Surber Square-foot or Hester-Dendy multiplate samples) will be submitted to the contractor on specified dates agreed upon with the Natural Resource Specialist of said Park (e.g. June, August, October). Total number of samples therefore will be agreed upon at beginning of each sampling year.

2) Preservation and handling: Gross, unpicked macroinvertebrate samples (Surber) will be preserved in a 5 percent formalin solution and stored in plastic containers. Samples will be (hand, UPS, US Mail) delivered to the contractor.

3) Sample analysis: The taxonomic references that will be used are to be listed. References previously used in park reports should be consulted.

Collembola: Order
Ephemeroptera: Genus/species when possible
Odonata: Genus
Plecoptera: Genus/species when possible
Trichoptera: Genus
Megaloptera: Genus/species when possible
Coleoptera: Genus
Diptera: Family/subfamily (Chironomidae)/Genus
Nematoda: Phylum
Annelida: Class
Amphipoda: Class
Acarina: Order
Gastropoda: Family
Pelecypoda: Family
Decapoda: Family
Turbellaria: Genus

4) Timeframe: Three copies of the final report will be submitted to the Natural Resource Specialist no later than 120 days from receipt of the samples.

5) Reports: The contractor will prepare a written report for each lot of samples which contains the following information:

- Actual count of each species/sample
- Species density estimate (quantitative samples)

6) Preservation and Storage of Macroinvertebrate Samples: Following identification to the lowest taxa level, specimens will be preserved in taxa groups by site in a 70 percent ethanol alcohol solution. Storage for the preserved specimens will be in 4 dram screw cap vials or 6 dram vials if the sample volume is larger vials will be provided by contractor. All samples are the property of the National Park Service. Upon completion of the identification process the

contractor will notify the National Park and establish a pickup date for the preserved specimens.

7) Quality control: The contractor is encouraged to remain in close communication with the contract representative. The study will be conducted according to the applicable regulations for Good Laboratory Practices as described by EPA (Federal Register 29 November 1983). The contractor shall stipulate by name who will be identifying the biota and performing sample analysis. For each person, the contractor will list his/her qualification credentials. Preprinted data sheets will be provided for taxonomic identifications and counts of organisms. The efficiency of removing organisms from debris (sorting or picking) will be checked by having every 20th sample sorted again by another person. For those samples, debris will not be immediately discarded and another person will examine the debris in the same manner that it was originally sorted. The names, addresses, and phone numbers of three references of past work will be provided for each person. The work referred to should be similar in nature to that included in this statement of work. A permanently bound notebook will be used for documenting the chain of custody. The chain of custody will involve tracking each sample from its collection to computer entry and final repository.

8) Bid pricing: Bids should be computed as a price for each sample and for all samples combined in a lot.

Community Level Endpoints

Community level change and community level metrics represent a current active area of research. Methods are changing. We have chosen several community level parameters that are interpretable and can be calculated with the aide of the program we have developed. Briefly we consider density of the organisms in the community, the total number of taxa present, the number of taxa present in three sensitive orders of Insecta, (called the *EPT index*), two calculations of community diversity, Simpson's D and Shannon Wiener H', and two measures of community similarity, Jaccard's coefficient (based on presence or absence of a taxa) and Pinkham and Persons B (a similarity coefficient considering taxa and their relative densities).

The number of qualitative EPT (Ephemeroptera/Plecoptera/Trichoptera) taxa or EPT index is the number of taxa in the three orders represented in a sample. Many species in these three orders are considered pollution-sensitive (Hawkes 1979). Surveying the number of EPT taxa over time and comparing the number with streams with of similar size and in close proximity may be indicative of present and past water quality.

The measurement of diversity has two components taxa richness and relative abundance. Richness is simply a count of the number of taxa (i.e. species) that are present. Relative abundance, or evenness, is a description of how the number of organisms in a community are distributed among the taxa. For monitoring and assessment within the parks three measures of diversity are used to describe benthic communities which are; taxa richness, Shannon-Wiener diversity index, and Simpson's index. One of the simplest and most basic measures used in aquatic ecology is taxa richness, which is simply the number of different taxa found over a given space and time. The Shannon diversity index and Simpson's index utilize both of the components of diversity and are based on the proportional abundance of species. A recent review of the use and meaning of ecological diversity may be found in Magurran (1988). Washington (1984) reviewed the ecological application of these diversity indices along with biotic and similarity indices.

The Shannon-Wiener diversity index is based on information theory and relates to the uncertainty of the identity of an individual chosen at random (Washington 1984, Magurran 1988). Shannon's index was calculated as:

$$H' = -\sum p_i \ln p_i$$

where p_i is the proportion of individuals in taxon i, or more specifically, $p_i = n_i/N$ where n_i is the number of individuals in the i^{th} taxon and N is the total number of individuals in the sample. The Shannon index is sometimes calculated with $\log_2$ or $\log_{10}$ rather than the natural log and the values obtained would differ by a constant (Brower and Zar 1977, Magurran 1988). This index is one of the most widely reported in ecological literature (Washington 1984).

Simpson (1949) proposed a formula that gives the probability that two individuals drawn at random from a finite community will belong to different taxa (Washington 1984, Magurran 1988). Simpson's Index was calculated as:

$$D = (\sum n_i(n_i - 1)) / N(N - 1)$$

where n_i is the number of individuals in the i^{th} taxon and N is the total number of individuals in the sample. The value of Simpson's index ranges from 0 to 1 and decreases with increasing diversity. Simpson's Index is most sensitive to the abundant species, and less weighted toward sample size or species richness (Magurran 1988).

Similarity or comparison indices are mathematical measures of the similarity of two community structures. An impacted community is usually compared to a reference site upstream or non-impacted site within the same ecoregion. Species composition, abundance, or both may be measured. These indices have been extensively used in plant ecology (Washington 1984, Hellawell 1986) and have recently been adapted for use with aquatic communities. Popular similarity indices include Pinkham and Pearson's index (1976) and Jaccard's index (1912).

PROCESSING AND ANALYSIS OF FIELD COLLECTED MACROINVERTEBRATES

Processing and analysis of field collected macroinvertebrate samples.

This section presents the methods for field collection of water chemistry and benthic macroinvertebrate samples using the Surber square foot bottom sampler. Additionally, it summarizes macroinvertebrate sample processing methods. Also included are a sample field checklist and data sheet, field and laboratory equipment list, a list of equipment suppliers, and a ~~benthic macroinvertebrate data sheet.~~

Sentinel Sites

Three permanent sampling sites have been established within the boundaries of WCNB benthic macroinvertebrate sampling and water chemistry monitoring. Two sites were established on Wilson Creek (Wilson Upper, Wilson Lower) and one on Skeggs Branch (Skeggs). Wilson Upper was located north of the bridge where the visitor's road first crosses Wilson Creek. The site was approximately 0.4 km from where the creek enters the National Battlefield. Wilson Lower was about 25 m south of the visitor road bridge at the second crossing of the creek. Skeggs was located on Skeggs Branch where the branch enters the western boundary of the National Battlefield, 10m from the Road ZZ bridge.

It is suggested that these sites be marked with distinct, long-lasting site markers. The following method to affix markers is modified from Voshell and Hiner (1990, Fig. 1). The markers consist of 0.9 m (3 ft) high, 5 cm (2 inch) diameter PVC pipe and are secured to 102 cm (40 inch) long, 1.3 cm (1/2 inch) diameter sharpened steel rebar. Before going to the field, drill a 1 cm (5/16 inch) hole in the PVC about 30 cm (12 inches) from the bottom. Drive the rebar about 2/3 the length into the ground and attach a chain link to the rebar with a hose clamp. Pass a 0.6 x 7.6 cm (1/4 x 3 inch) hexhead bolt through the chain link and pre-drilled holes in the PVC pipe. Finally, attach an end cap to the PVC pipe. For greatest visibility, we suggest the PVC markers be white with black end caps.

Water Chemistry

Water samples should be collected before the stream is disturbed by macroinvertebrate collecting. Collect samples in an area of flowing water representative of the sample reach. Fill bottles by leaning from shore or standing in stream while leaning upstream so as not to disturb the stream bottom. Hold the bottle slanting it slightly upstream to reduce the possibility of contamination by the operator. Many water quality samples need to be kept as cool as possible to protect their integrity. Additionally, other methods of sample preservation may be desired, including chemical addition or filtering. Consult with the laboratory doing the chemical analysis for preferred sampling methodology and sample preservation. After water samples have been taken, in situ measures, including temperature and pH, should be conducted. Take these measures in moving water, out of direct sunlight.

Benthic Macroinvertebrates

Macroinvertebrates are generally considered those invertebrates, such as worms, mollusks, and arthropods, large enough to be seen with the unaided eye (Weber 1973). However, the very early stages of these organisms are often only detectable with the aid of a stereomicroscope or magnifier. The term benthic refers to organisms living on the bottom of aquatic environments or on firm substrates protruding above the bottom. Benthic faunal communities usually contain a wide variety of organisms. Many of the community members are the immature stages (nymphs and larvae) of insects which leave the water for a terrestrial adult stage. During this terrestrial period, reproduction and dispersal take place.

Quantitative benthic sampling is accomplished with a Surber bottom sampler (Merritt et al.

1984, Fig. 2). Five replicate Surber samplers should be taken at each sampling site from riffle habitats. The placement of the sampler is limited by stream depth, current, and substrate. The depth and current must be sufficient to dislodge and carry organisms into the catch net, but the sampler should not be completely submerged. The substrate must be regular enough so organisms will not be washed under the sampler.

In addition to the sampler, the equipment needed for sampling are a vegetable brush, a small hand rake (garden cultivator), a wash bottle, and forceps. Sample labels, preservative, and Zip-loc bags are needed for sample storage. Plastic labeling tape (such as Dymo brand) with the back still attached makes an effective permanent label.

Five quantitative samples should be taken in riffle habitats with in the sample reach. Take samples in different segments of the reach, always begin downstream and walk upstream to find the next suitable sampling area. Benthic macroinvertebrates are mobile, and measurements of their abundance will be affected by walking through areas to be sample. After placing the sampler on the bottom, check to make sure that the frame of the sampler makes good contact with the substrate. The best position for the person taking the sample is to kneel or crouch behind the sampler with the catch net passing between the legs. A second person to assist in holding down the sampler may be desirable. Brush each individual rock on all sides with the vegetable brush, so that the organisms will be dislodged and swept into the catch net. This is best done by holding the rocks underwater to make sure that no organisms are thrown out of the sampler. Each rock should also be visually examined at close range, because many aquatic insects have special means of attaching themselves very tightly to rock surfaces. Use forceps to remove any organisms found clinging after brushing. After all of the larger rocks have been brushed, examined and removed, rake the remaining fine substrate to stir up the sediment inhabiting organisms. Try to rake down to a depth of about 8-10 cm.

The catch net is now washed several times to concentrate the contents into the end. This is best accomplished by raising the sampler out of the water, then briefly submersing the net raising it rapidly. Splashing water along the sides of the net is also effective. The contents of the sample are placed into a plastic Zip-lock bag by inverting the net. It is usually necessary to invert and rewash the net several times to get all of the contents into the bag. Rinse any remaining organic matter into the sample bag with a wash bottle. Visually inspect the catch net, pick off any invertebrates with forceps and place them in the bag. The appropriate label (site, date, replicate) should be placed in the bag immediately. Add an appropriate preservative. After preservative is added, squeeze bag to let as much air out as possible before sealing the bag. To insure integrity of sample, place this sealed bag into another bag. If the samples are to be processed within 24 hours they may be placed on ice, to be kept as near freezing as possible in lieu of adding preservative.

Several fluids are commonly used to preserve benthic samples including formaldehyde, ethanol and isopropyl alcohol. Formaldehyde is recommended for benthic samples because of the bacterial load of the detritus and sediment in the samples. However, there are health concerns associated with formaldehyde, so it must be used with caution. The final dilution of formaldehyde should be 5% of the standard stock solution. The standard stock solution, sometimes called formalin, contains about 37% formaldehyde, therefore, the final concentration of formaldehyde in the sample is approximately 2% (Voshell and Hiner 1990). To preserve the sample with a formaldehyde solution, add some stream water to the bag with the sampler and add enough preservative to equal about 5% of the liquid.

Ethanol and isopropyl alcohol are adequate substitutes for formaldehyde. Use a 70% solution of these alcohols. Special attention must be given to dilution when preserving with alcohol. Lower concentrations are not adequate to retard bacterial decomposition, and higher concentrations make specimens brittle. We recommend that when preserving by this method, if the samples are to be stored for longer than a week, the alcohol be replaced by laboratory diluted solution of 70% alcohol to ensure protection from bacterial decomposition.

Qualitative sampling of macroinvertebrates may be desired at this time. This sampling is conducted to supplement the species list. In order to obtain adult stages of benthic insects, an aerial net should be used to sweep the riparian vegetation up and down the stream banks. Selected specimens may be retained in a small bottle with 70-80% ethanol and at least a temporary site label. A standard D-frame kick net may be used to obtain specimens from stream microhabitats not sampled with the quantitative sampler. Hold net downstream and dislodge organisms by hand or by kicking. Common microhabitats to sample include leaf packs, underneath log, underneath large rocks, and by exposed roots and vegetation. As with the aerial net sampling, preserve selected specimens in a small bottle with 70-80% ethanol. Additionally, mature specimens may be returned to a laboratory for rearing to adult stages if facilities are available.

Sample Processing

Store bags with Surber samples in airtight containers to reduce escape of formaldehyde fumes. The processing steps for benthic macroinvertebrate samples are 1) washing, 2) sorting, and 3) identification and enumeration. It is important to keep the sample label with the sample through all the steps, and keep a log that includes the date each step was completed, the initials of the person completing the step, and any notes.

The first step in sample analysis is to wash organisms and formaldehyde preservative from the sampler. First, rinse the sampler over a fine mesh sieve with tap water. A U. S. standard No. 60 sieve (sieve openings 0.25 mm) was used in this study. Others recommend sieves with larger openings, such as No. 30 (sieve openings 0.6 mm, Weber 1973). A finer sieve retains early instars while a coarser sieve reduces sample volume and therefore hastens processing. Gently stir the sample and shake the sieve under the water to clean the sample without damaging the delicate specimens. Transfer the sample from the sieve into a glass beaker. Cover the sample with water if it is to be picked immediately, or add 70% ethanol if not. Always keep the original sample tag with the sample.

Picking the organisms from the sample, or sorting, is the next step. Into the bottom of a clear petri dish, pour enough of the sample to cover the bottom of the dish. Under a binocular dissecting microscope at low power (about 10X) separate the invertebrates from the debris and place the specimens in a vial with 70-80% ethanol. Patent lip vials with neoprene stoppers or screw cap vials with a polyethylene liner are both adequate for storage of samples and reducing evaporation of alcohol. The 7 g (4 dram, usually 21 X 70 mm) size vials are a good size for storage of samples. Look through the petri dish in a systematic manner, and then scan the dish again to check the work. Depending on the expertise of the person sorting the sample, different taxa may be separated into a sorting tray or separate vials. It is imperative that every vial have a label. If paper is used for the label it should have at least a 50% rag content, preferably 90-100%, and written with #2 pencil or India ink.

The final step in processing the sample is identifying the organisms to the lowest practical taxonomic level and counting the number in each taxon. Each specimen needs to be examined with a good quality binocular dissecting microscope. Taxonomic publications with descriptions and keys are utilized to make identifications. Merritt and Cummins (1984) thoroughly treats all the aquatic and semiaquatic insect orders. More specialized works for individual orders include Edmunds et al. (1976), Ephemeroptera (mayflies); Stewart and Stark (1989), Poulton and Stewart (1991), Plecoptera (stoneflies); and Wiggins (1977), Trichoptera (caddisflies). A good reference for macroinvertebrates other than insects is Pennak (1989). Benthic macroinvertebrates should be identified to at least the following taxonomic levels:

Collembola: Order
Ephemeroptera: Genus/Species
Plecoptera: Genus/Species
Hemiptera: Genus
Megaloptera: Genus
Trichoptera: Genus
Lepidoptera: Genus
Coleoptera: Genus
Diptera: Family/Subfamily (Chironomidae)/Genus
Nematoda: Phylum
Turbellaria: Genus
Annelida: Class
Acarina: Class
Isopoda: Genus
Amphipoda: Genus
Decapoda: Family
Gastropoda: Family
Pelecypoda: Family

It may not be possible to identify early stages of macroinvertebrates to these levels, and with some insect genera, adult specimens are needed for identification. Note that taxonomic references are written for people who are already familiar with the taxonomic group covered. Formal training with a specialist is necessary for accurate identifications. It is also a good idea to have representative specimens verified by specialists. A data sheet that lists many of the species encountered during benthic sampling of the Wilson Creek and Skeggs Branch in **WCNB** is included. The list is meant for use as a general guide and does not list all species or all life stages that may be encountered.

Each sample may be stored in a single vial or several vials, separated by taxonomic group. Every vial needs to have an identifying label which include the site, date and replicate. The original field label should be kept with the corresponding sample. Inevitably, through the data entry and analysis process, questions will be raised and specimens may need to be reexamined. Cardboard unit trays are available for vial storage. The **WCNB** must decide how long to keep the samples after identification, data entry, and data analysis. A minimum of one year after data analysis is recommended. It is helpful to keep several specimens of each taxon collected each year as a voucher collection. This voucher collection can serve as a tool to answer taxonomic questions, train employees or consultants, and may also be used as an educational tool (Voshell and Hiner 1990).

Figure 1. Streamside marker (Voshell and Hiner 1990).

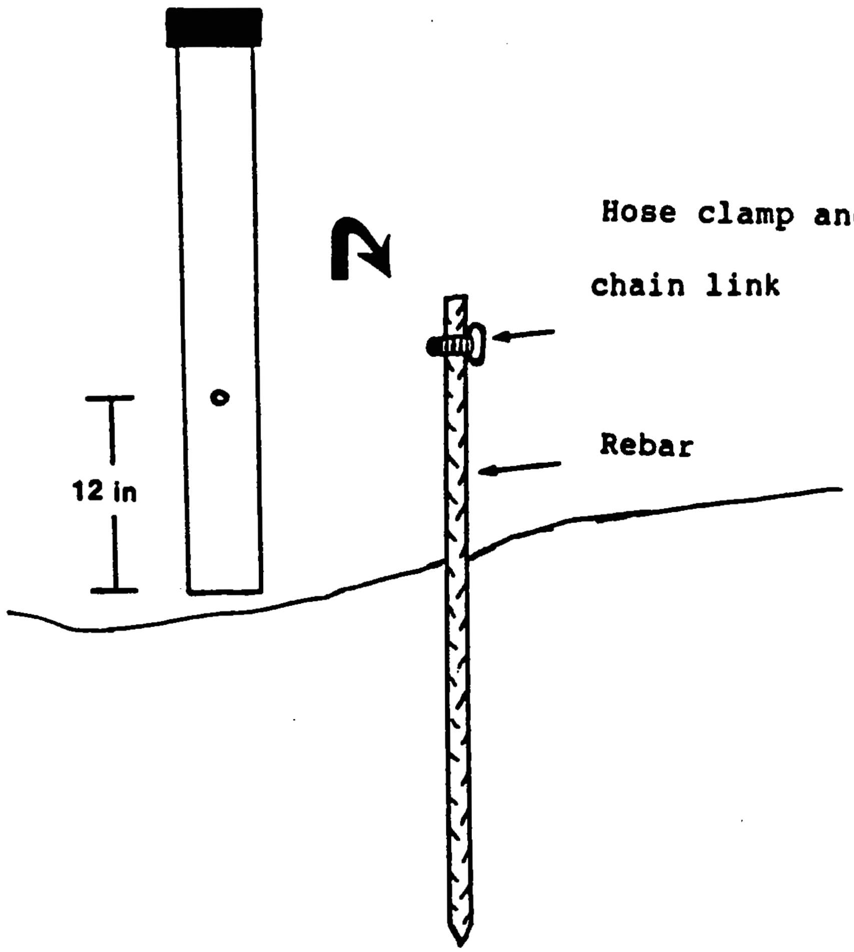
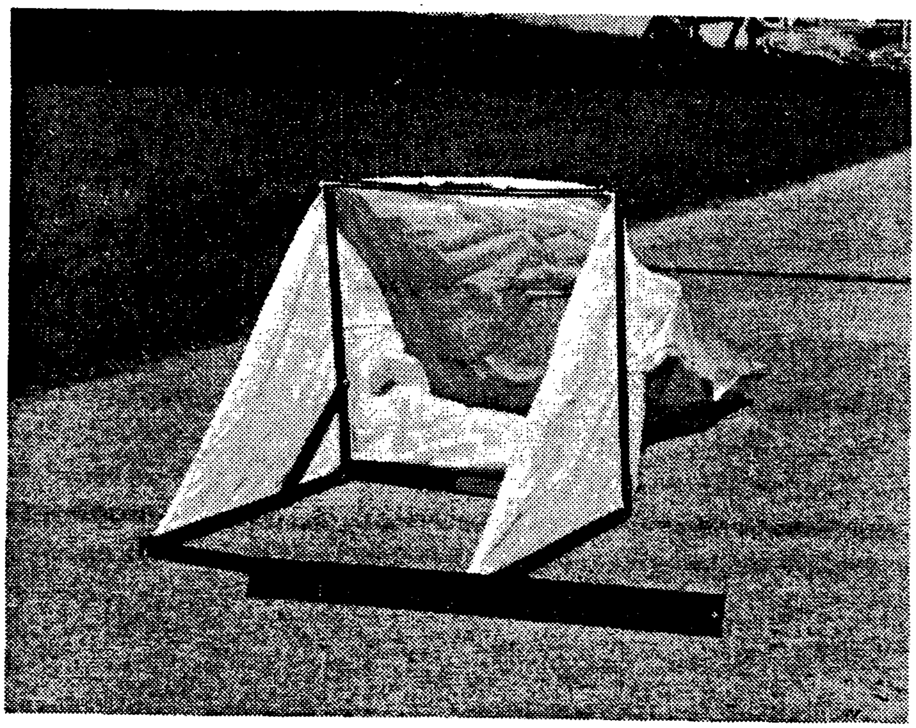


Figure 2. Surber bottom sampler (Merritt et al. 1984).



Field checklist and data sheet.

FIELD RECORDS

Site ____________________ Date ____________________

Workers ____

Air temp. ____________°C

H_2O temp. ____________°C

pH _____________ Method: ____________________________

Water Samples: ____

Surber Samples: ____

All samples labeled? _____

Notes: ____

____

____

____

____

Field and laboratory equipment list.

I. General Field

_____ Hip Waders
_____ Field Notebook
_____ Pencil

II. Water Chemistry Sampling

_____ Water bottles for samples
_____ Cooler with ice left in vehicle
_____ pH meter or Hellige colorimeter
_____ Thermometer

III. Benthic Macroinvertebrate Sampling

Quantitative sampling
_____ Surber sampler
_____ Vegetable brush
_____ Small hand rake (garden cultivator)
_____ Wash bottle
_____ forceps
_____ Zip-loc bags (10 per site)
_____ Sample labels (Site, Date, Replicate)
_____ Preservative (Formaldehyde recommended)

Qualitative sampling
_____ Aerial net
_____ D-frame kick net
_____ enamel pan
_____ small bottles with ethanol
_____ rag paper for labels

IV. Macroinvertebrate Sample Processing

____ Binocular dissecting microscope
____ Sieve
____ Ethanol
____ Beakers
____ Petri dishes
____ Fine forceps
____ Sorting tray
____ Vials for storage
____ Cardboard unit trays
____ Tally counter

List of vendors of selected equipment needed for field and laboratory procedures. There are alternative vendors for most items.

Item	Vendor
Polyethylene or glass, water sample collection bottles	Baxter Scientific 1118 Clay St. Kansas City, MO 64116 800-892-2433 (MO) 800-821-2206 (other than MO)
pH meter	Baxter Scientific
Surber sampler	Wildco 301 Cass St. Saginaw, MI 48602 517-799-8100
Aerial net, D-frame kick net	Bio Quip Inc. 17803 LaSalle Ave. Gardena, CA 90248 213/324-0620
Glass vials	Bio Quip Inc.

Macroinvertebrate data sheet. Includes many of the species encountered during benthic sampling of the **WCNB**. This list is meant for use as a general guide and does not list all species or all life stages that may be encountered. Taxon refers to larvae, unless noted.

BENTHIC MACROINVERTEBRATE DATA

Site ________ Sample Date ____________ Replicate ________

Identifier _______________________

Taxa	No.	Comments
Collembola		
Ephemeroptera		
Baetidae		
Acentrella sp.		
Baetis flavistraga		
Diphetor hageni		
Caenidae		
Caenis sp.		
Ephemerellidae		
Eurylophella aesitiva		
Heptageniidae		
Stenacron interpunctatum		
Stenonema femoratum		
Leptophlebiidae		
Paraleptophlebia sp.		
Tricorythidae		
Tricorythodes sp.		
Odonata		
Coenagrionidae		
Argia spp.		
Calopterygidae		
Calopteryx sp.		
Hetaerina sp.		
Plecoptera		
Leuctridae		
Zealeuctra sp.		
Nemouridae		
Amphinemura sp.		
Perlidae		
Acroneuria evoluta		
Agnetina flavescens		
Neoperla falayah		
Perlesta balukta		
Perlinella ephrye		
Perlodidae		
Clioperla clio		

Hemiptera		
Gerridae		
Gerris sp.		
Veliidae		
Paravelia sp.		
Microvelia sp.		
Rhagovelia obesa		
Megaloptera		
Corydalidae		
Corydalus cornutus		
Sialidae		
Sialis sp.		
Trichoptera		
Hydropsychidae		
Cheumatopsyche sp.		
Hydropsyche sp.		
Hydroptilidae		
Hydroptila spp.		
Hydroptila spp. (Pupae)		
Neotrichia sp.		
Ochrotrichia spp.		
Ochrotrichia spp. (Pupae)		
Oxyethira spp.		
Leptoceridae		
Ceraclea transversus		
Oecetis inconspicua		
Philopotamidae		
Chimarra spp.		
Phryganeidae		
Ptilosomis ocellifera		
Polycentropodidae		
Polycentropus sp.		
Psychomyiidae		
Psychomyia flavida		
Lepidoptera		
Pyralidae		
Petrophila sp.		
Coleoptera		
Elmidae		
Dubiraphia sp.		
Dubiraphia sp. (Adults)		
Optioservus sp.		
Optioservus sp. (Adults)		
Stenelmis spp.		
Stenelmis spp. (Adults)		
Gyrinidae		
Dineutus sp.		
Hydrophilidae		
Berosus sp.		

Psephenidae		
Ectopria sp.		
Psephenus sp.		
Diptera		
Ceratopogonidae		
Chironomidae		
Chironomidae (Pupae)		
Chironominae		
Chironomini		
Tanytarsini		
Orthocladiinae		
Tanypodinae		
Diamesinae		
Dixidae		
Dixa sp.		
Empididae		
Hemerodromia sp.		
Hemerodromia sp. (Pupae)		
Simuliidae		
Simulium sp.		
Simulium sp. (Pupae)		
Stratiomydae		
Tipulidae		
Antocha sp.		
Antocha sp. (Pupae)		
Tipula spp.		
Tipula spp. (Pupae)		
Nematoda		
Annelida		
Oligochaeta		
Hirudinea		
Turbellaria		
Planariidae		
Dugesia		
Isopoda		
Asellus sp.		
Lirceus sp.		
Amphipoda		
Gammarus sp.		
Hyalella sp.		
Decapoda		
Orconectes virilis		
Acarina		
Gastropoda		
Anyclidae		
Physidae		
Planorbidae		
Pelecypoda		
Sphaeriidae		

MACROINVERTEBRATE COMMUNITY

ANALYSIS

Data Analysis

In order to facilitate in the analysis of macroinvertebrate data collected by the park a dBase III+ computer program, National Parks Sample Collection (**NAPSAC**; Krause 1991) was developed. NAPSAC is a menu driven program that assists in data entry, storage, and classification of macroinvertebrate data. Output from the NAPSAC program is designed to interface with BSTRAP (Mouser 1991). BSTRAP is a community analysis program that computes both Simpson's D and Shannon Wiener H' diversity indices along with assessing similarities between communities using Pinkham and Pearson's B and Jaccard's Index. For each index, a bootstrapping estimate of standard error is computed, and confidence intervals given.

Operation Guide to NAPSAC and BSTRAP

Getting Started:

The first thing that must be done is to copy NAPSAC, BSTRAP, and the preloaded DATA disks into a dBase subdirectory. To do this, go to your dBase directory by typing

C:**CD\DBASE**

press [RETURN] and your prompt should appear as

C:\DBASE>

From this directory, create a subdirectory called NAPSAC by typing

C:\DBASE>**MD NAPSAC**

then press [RETURN]. To get "into" that subdirectory type,

C:\DBASE>**CD NAPSAC**

Now your prompt should appear as

C:\DBASE\NAPSAC>

Copy the NAPSAC, BOOTSTRAP, and preloaded DATA disks (if data available) into this subdirectory by inserting the disk containing NAPSAC into the disk drive and typing

C:\DBASE\NAPSAC>COPY A:*.*

and press [RETURN], when this is finished, exchange the NAPSAC disk with the BOOTSTRAP disk and repeat, then do the same with the DATA disk. (NOTE: If you are entering a preloaded data disk it must be copied last, otherwise data dictionaries will be over written).

dBase path

You will need to add dBase to the AUTOEXEC.BAT file in your root directory, or to the path of the menu driven program. Consult your Dos or menu program manual for instructions. Also check the NAPSAC.BAT file that has been copied into the subdirectory which contains the following commands:

```
ECHO
REM This file will start DBASE III+ using the DBASE3\NAPSAC
REM Subdirectory. This is the development area for the
REM Natural Parks Entomology Study system.
CLS
C:
CD\DBASE\NAPSAC
DBASE
```

The Drive on line 6 and directories on lines 7 and 8 must match the drive and directories you are currently using. Using edlin (consult Dos manual) change these lines to match your system.

NAPSAC

To start the NAPSAC system, at the DOS prompt type:

C:\DBASE\NAPSAC>NAPSAC

and the system will begin loading.

The first screen displayed is the Ashton-Tate licensing agreement. Pressing [RETURN] will erase the screen and start the NAPSAC system. After the licensing agreement screen clears, the NAPSAC welcome screen will be displayed:

```
WELCOME TO THE NATIONAL PARKS NAPSAC SYSTEM

This system is for use by the United States
National Park Service.  It is designed to
track the Benthic Macroinvertebrate community
information collected at park sites.
The system was written by Don Krause of
FTL technologies, and is meant for the sole
use of the United States Government and its'
agencies.  Any questions concerning this
product should be directed to the National
Park office at Colorado State University.
No warranties, expressed or implied are made
concerning this product.

    Press any key to start the system.
```

MAIN MENU

Press any key and the NAPSAC Main menu will be displayed. The Main menu is the core of the system. From it, the operator may select to maintain the Sample file, the Taxa Dictionary file, print reports, or run the monthly validation program. Each of these sections are discussed in detail in other parts of the documentation. An example of the Main menu is displayed below:

```
———————— NATIONAL PARKS NAPSAC - SAMPLE COLLECTION SYSTEM ————————

                         MAIN MENU

            1 -  MAINTAIN SAMPLE DATA
            2 -  MAINTAIN TAXA DICTIONARY DATA
            3 -  REPORTS AND OUTPUT



            9 -  MONTHLY VALIDATION RUN
            Q -  QUIT NAPSAC SYSTEM

     PLEASE ENTER YOUR CHOICE: _
```

Enter either a 1, 2, 3, 9, or Q, and then press [RETURN]. If any other key is pressed, the menu will be redisplayed and a valid choice will need to be entered. When making menu selections in this system, the [RETURN] key must always be pressed.

If a 1 is entered, the Sample menu will be displayed. From the Sample menu, Sample records may be Added, Changed, Displayed, or Deleted.

If a 2 is entered, the Taxa Dictionary menu will be displayed. From the Taxa Dictionary the operator may Add, Change, Display, or Delete records in the Taxa Dictionary file.

If a 3 is entered, the Report menu will be displayed. From the Report menu, several reports may be generated. The operator may also generate an ASCII file of the sample records. This file can be loaded into the BSTRAP program for analysis.

If a 9 is entered, the system will run a program that validates the taxa codes in the Sample database. If checks to be sure that all the taxa codes are in the Taxa Dictionary file. If it finds any invalid taxa codes, it displays an error message. The list of invalid taxa codes can be printed using a choice in the Report menu.

If a 'Q' is entered, the NAPSAC system will end and the operator will be returned to the DOS prompt. It is necessary to always return to DOS before turning off the computer. If the computer is turned off while the NAPSAC system is still running, data that was entered during that session may be lost.

SAMPLE MAINTENANCE MENU

If a '1' is selected from the Main menu, the Sample Maintenance menu will be displayed. From this menu, Sample records may be added, changed, displayed, or deleted:

```
———— NATIONAL PARKS NAPSAC - SAMPLE COLLECTION SYSTEM ————

                       MAIN MENU

              1 -  MAINTAIN SAMPLE DATA
              2 -  MAINTAIN TAXA DICTIONARY DATA
              3 -  REPORTS AND OUTPUT

              SAMPLE MAINTENANCE SUB MENU

     A -  ADD NEW SAMPLE RECORD TO FILE
     C -  CHANGE EXISTING SAMPLE RECORD
     D -  DISPLAY EXISTING SAMPLE RECORD
     X -  DELETE EXISTING SAMPLE RECORD
     Q -  QUIT, RETURN TO MAIN MENU

   PLEASE ENTER YOUR CHOICE: _
```

When maintaining the Sample file, the operator will enter the Site, Date, Replicate, Taxa Code, and number of occurrences of the Taxa. NOTE: **Samples collected during the 1989 and 1990**

sample seasons have been pre entered (see SAMPLE RECORD output). For each selection on the Sample Maintenance menu, the following Sample Maintenance screen will appear:

```
National Parks NAPSAC System
 Sample Maintenance Screen

_______ Mode
                                                              Number
Site: ____  Date: __/__/__  Replicate: __  Taxa Code: ____  Found: ____

Enter a "9" in SITE to Exit without updating the file

     Order: ____________________      Family: ____________________
     Genus: ____________________
   Species: ______________________________________
Life Stage: _                     Funct Group: __

                         Messages:
```

When the screen is displayed, all the fields, except Mode will be blank. The mode field will say either Add, Change, Display, or Delete depending on which option was chosen from the Sample Maintenance menu.

At any time, the operator may enter a '9' in the Site field to exit the current function and return to the Sample Maintenance menu. The operator may enter information in the Site, Date, Replicate, Taxa Code, and Occurrence fields. When the Taxa Code field is filled, the program will read the Taxa Dictionary file to retrieve the Order, Family, etc. and display that information on the screen. The Message area is used to send messages to the operator.

The following is a list of each of the fields that the operator may fill and a description of the data entry requirements for the fields:

SITE: This is the Site at which the sample was taken. It may be from one to five characters in length. Any characters may be entered. Letters will be automatically converted to upper case. This field must be entered.

DATE: This is the Date when the sample was taken. It must be entered in MM/DD/YY format. The /'s do not need to be entered. If the month or day is only one digit, a '0' must precede the entry. For example, January 8, 1991 would be entered: 01/08/91. If the date is entered incorrectly, the cursor will move to the beginning of the Date field and the operator may reenter the date. This field must be entered.

REPLICATE: This is the Replicate at which the sample was taken. It can be a number from one to 99. This field must be entered.

TAXA CODE: This is the Taxa Code for the organism. It can be from one to four characters. The code will be verified against the Taxa Codes in the Taxa Dictionary file (see Taxa Dictionary output). Any characters will be automatically converted to upper case. If the Taxa Code is not found in the dictionary, then an error message will be displayed. The cursor will reappear in the Taxa Code field and the code can be corrected. If the Taxa Code is valid, but not in the Taxa Dictionary, before this entry can be recorded, the operator must go to the Taxa Dictionary Maintenance screen and add the Taxa Code to the dictionary. This field must be entered.

After the Site/Date/Rep/Taxa (SDRT) are entered, the system will check to see if that combination already exists on the system.

If the operator is performing an Add, and the SDRT already exists, an error message will be displayed.

If the operator is performing a Change, Display, or Delete, and the SDRT does not exist, an error message will be displayed.

After all the fields have been entered, the system will ask you to verify that the information is correct. If not, enter a 'N', and you will be returned to the Site field.

ADDING RECORDS: When the operator is adding records to the Sample file, all the fields can be entered. The data will be validated as per the field descriptions above.

CHANGING RECORDS: When the operator is changing records, the only field may be changed is the Occurrence. Enter the Site, Date, Replicate, and Taxa. The Occurrence field will automatically be displayed with the information on file for that SDRT combination. Enter the correct number of occurrences. If any of the other fields need to be changed, it is necessary to Delete the current record and then Add the correct one. This is necessary because the Sample system is based on the SDRT combination.

DELETING RECORDS: If you wish to delete a sample record, enter the Site, Date, Replicate, and Taxa. The Occurrence will automatically be displayed. The computer will ask you to verify that you want to delete the record. If so, enter 'Y', otherwise 'N'. Once a sample record has been deleted, there is no way to retrieve it. If a record is accidently deleted, it may be reentered through the Add mode.

DISPLAYING RECORDS: This option allows the operator to examine one SDRT record without being able to change any information. Enter the Site, Date, Replicate, and Taxa. The Occurrence and other information will be displayed. When you want to leave this record, press any key.

TAXA DICTIONARY MAINTENANCE MENU

If a '2' is selected from the Main menu, the Taxa Dictionary Maintenance menu will be displayed. From this menu, Taxa Dictionary records may be added, changed, displayed, or deleted:

```
———————— NATIONAL PARKS NAPSAC - SAMPLE COLLECTION SYSTEM ————————

                         MAIN MENU

              1 -  MAINTAIN SAMPLE DATA
              2 -  MAINTAIN TAXA DICTIONARY DATA
              3 -  REPORTS AND OUTPUT

           TAXA DICTIONARY MAINTENANCE SUB MENU

     A -  ADD NEW SAMPLE TAXA TO THE DICTIONARY
     C -  CHANGE EXISTING TAXA RECORD
     D -  DISPLAY EXISTING TAXA RECORD
     X -  DELETE EXISTING TAXA RECORD
     Q -  QUIT, RETURN TO MAIN MENU

  PLEASE ENTER YOUR CHOICE: _
```

When maintaining the Taxa Dictionary file, the operator will enter the Taxa Code, Order, Family, Genus, Species, Life stage, and Functional Group. **NOTE: A Taxa Dictionary has been**

entered based on species occurrence data collected during the 1989 and 1990 macroinvertebrate collection season (see Taxa Dictionary output). For each selection on the Taxa Dictionary Maintenance menu, the following Taxa Dictionary Maintenance screen will appear:

```
                    National Parks NAPSAC System
                    Taxa Dict Maintenance Screen

_______ Mode

Taxa: ____      Order: ____________________
               Family: ____________________
                Genus: ____________________
              Species: ________________________________________
           Life Stage: _

Enter a "9" in TAXA to Exit without updating the file

                          Messages:
```

When the screen is displayed, all the fields, except Mode will be blank. The mode field will say either Add, Change, Display, or Delete depending on which option was chosen from the Taxa Dictionary Maintenance menu.

At any time, the operator may enter a '9' in the Taxa field to exit the current function and return to the Taxa Dictionary Maintenance menu. The operator should enter the Taxa Code in the Taxa field. When the Taxa is entered, if the code is already in the Taxa Dictionary file, the other fields will be filled automatically. The operator may change any of the fields on the screen. The Message area is used to send messages to the operator.

The following is a list of each of the fields that the operator may fill and a description of the data entry requirements for the fields:

TAXA: This is the Taxa Code for the organism. It can be from one to four characters. Any characters will automatically be converted to upper case. This field must be entered.

ORDER: This is the Order in which the organism belongs. It can be from one to twenty characters. This field must be entered.

FAMILY: This is the Family in which the organism belongs. It can be from one to twenty characters. This field must be entered.

GENUS: This is the Genus in which the organism belongs. It can be from one to twenty characters. This field must be entered.

SPECIES: This is the Specific name of the organism. If can be from one to forty characters. Entry into this field is optional.

LIFE STAGE: This is the Life Stage code for the organism. It is one character long and will automatically be converted to upper case.

FUNCTION GROUP: This is the Functional Group code of the organism. It can be either one or two characters and will automatically be converted to upper case.

When you enter the Taxa Code, the system will check to see if it already exists in the Taxa Dictionary.

If the operator was performing an Add, and the Taxa Code already exists, an error message will be displayed.

If the operator was performing a Change, Display, or Delete, and the Taxa Code does not exist, an error message will be displayed.

After all the fields have been filled the system will ask you to verify that the entries are correct. If they are not, enter a 'N' and you will be returned to the Taxa field.

ADDING RECORDS: When the operator is adding records to the Taxa Dictionary file, all the fields can be entered. The data will be validated as per the field descriptions above.

CHANGING RECORDS: When the operator is changing records, all the fields may be changed except the Taxa Code. Enter the code of the Taxa whose information you wish to change. The other fields will be displayed with the information currently on file. Use the [RETURN] key to move to the field you wish to change and correct the field. Move through the rest of the fields. When you pass the last field, the system will ask you if the information you entered is correct. If you wish to change a Taxa Code, you must Delete the old Tax Dictionary record, and Add the new one.

DELETING RECORDS: If you wish to delete a Taxa Code from the Taxa Dictionary, enter the code. The other fields will be displayed with the information currently on file. The computer will ask you to verify that you want to delete the record. If so, enter 'Y', otherwise 'N'. Once a Taxa Dictionary record has been deleted, there is no way to retrieve it. If the record is accidently deleted, it may be reentered through the Add mode. When you delete a Taxa Dictionary record, It is recommended that you run the Monthly Validation Run (option 9 on the Main menu). This will report if any Samples have the Taxa Code that was deleted. If so, then the Taxa Code will need to be reentered in the dictionary.

DISPLAYING RECORDS: This option allows the operator to examine one Taxa Dictionary record without being able to change any information. Enter the Taxa Code. The other fields will be displayed with the information currently on file. When you want to leave this record, press any key.

MONTHLY VALIDATION REPORT

This program is used to verify that the Taxa Codes in the Sample file have a corresponding Taxa Code in the Taxa Dictionary. If there are any errors, the report can be printed using option 'E' in the Report menu.

When entering records into the Sample file, the system verifies that the entered Taxa Code exists in the Taxa Dictionary. Therefore, it is not possible to enter an invalid Taxa Code in the Sample file. However, it is possible to delete a Taxa Code from the Taxa Dictionary, and a record in the Sample file will still have that Taxa Code. If this occurs, the statistical reports generated through BSTRAP will be incorrect. This program will eliminate that potential problem.

It is recommended that this program be run at least once per month. Two other times when it should be run are:

1) Whenever a record is deleted from the Taxa Dictionary.

2) Immediately before running option 'P' of the Report menu.

(Output ASCII file to the BSTRAP system)

When the program is completed, a message will be sent to the screen letting the operator know if there where any errors found. An error is a Taxa Code found in the Sample file but not in the Taxa Dictionary.

If there are no errors, then no action needs to be taken. If there are errors, then go to the Report menu and print the Validation Error report (option E). The Taxa Codes listed on the report will need to be reentered into the Taxa Dictionary file.

REPORT MENU

If a '3' is selected from the Main menu, the Report menu will be displayed:

```
NATIONAL PARKS NAPSAC - SAMPLE COLLECTION SYSTEM

   STANDARD REPORTS                 FILE DUMPS

A- SITE/DATE/REP LIST           J- SAMPLE            P- OUTPUT ASCII FILE
B-                              K- TAXA DICT         R-
C-                              L-                   S-
D-                              M-                   T-
E- VALIDATION ERROR REPORT      N-
F-                              O-
G-
H-
I-

          ENTER "Q" TO RETURN TO THE MAIN MENU

          PLEASE ENTER YOUR CHOICE: _
```

OUTPUTS

A - SITE/DATE/REP LIST

This report is a catalog of all the Site, Date, Replicate combinations in the Sample file. For each Site/Date combination, one line will be printed. Then, each Replicate that exists for that Site/Date will be listed next. For example:

SITE	DATE	REPLICATES:
CAT	01/01/91	1 2 3 4 5
WIL	01/01/91	1 2 3
ZAP	01/01/91	1 2 3 4

The list will be printed in Site + Date order.

E - VALIDATION ERROR REPORT

This is the error report generated by the Monthly Validation Run on the Main menu (option 9). The report lists the Site, Date, Replicate, Occurrences, and a message for each error found by the Monthly Validation Run. At this time, there is only one message: the Taxa code in the Sample record is not in the Taxa Dictionary file. This report can be sent to either the screen(S) or the printer(P).

J - SAMPLE FILE DUMP

This report is a list of all or some of the records is the Sample file. It lists the Site, Date, Rep, Taxa, and number of occurrences. The program will ask whether the operator wants to send the report to the screen (S) or the printer (P). The program then asks the operator which records to list. It asks for the Site, Date, and Replicate to print. If you want all the records on file, just press return in the Site, Date, and Replicate field. If you want all the records for a particular Site, enter the Site and leave the other fields blank. If you want all the records for a particular Site, on a particular date, enter the Site and Date, and leave

Replicate blank. Any combination of Site, Date, and Replicate can be entered.

K - TAXA DICT FILE DUMP

This report is a list of all the records in the Taxa Dictionary. It lists the Taxa Code, Order, Family, Genus, Species, Life Stage, and Functional group. The report can be printed alphabetically by either Taxa Code(T), or Order/Family(O). The program will ask for the order when it starts. This report can only be sent to the printer.

P - OUTPUT ASCII FILE

This program will create an ASCII file (**SAMPOUT.ASC**) that can be loaded into the bootstrapping BSTRAP system. When the operator runs this program, it will ask for the Beginning and Ending dates for the samples to output. Enter the dates in MM/DD/YY format. For example January 15, 1991 would be entered: 01/15/91. Notice the leading zero for January. Any dates that are less than 10, must have a leading zero entered. If you wish to create more than one output file you must exit NAPSAC and rename the SAMPOUT.ASC file created before creating a second output file (consult DOS manual for RENAME command).

After creating your output ASCII file return to the main menu and exit the NAPSAC program. This will return you to the NAPSAC subdirectory within dBase. At the prompt type

C:\DBASE\NAPSAC>**BSTRAP**

to start the bootstrapping program.

SITE	DATE	REPLICATES				
LOWER	08/15/88	1	2	3	4	5
LOWER	10/14/88	1	2	3	4	5
LOWER	04/17/89	1	2	3	4	5
LOWER	06/26/89	1	2	3	4	5
LOWER	08/15/89	1	2	3	4	5
LOWER	10/25/89	1	2	3	4	5
LOWER	04/07/90	1	2	3	4	5
LOWER	06/27/90	1	2	3	4	5
SKEGS	08/15/88	1	2	3	4	5
SKEGS	10/14/88	1	2	3	4	5
SKEGS	06/26/89	1	2	3	4	5
SKEGS	08/15/89	1	2	3	4	5
SKEGS	10/25/89	1	2	3	4	5
SKEGS	04/07/90	1	2	3	4	5
SKEGS	06/27/90	1	2	3	4	5
UPPER	08/15/88	1	2	3	4	5
UPPER	10/14/88	1	2	3	4	5
UPPER	04/17/89	1	2	3	4	5
UPPER	06/26/89	1	2	3	4	5
UPPER	08/15/89	1	2	3	4	5
UPPER	10/25/89	1	2	3	4	5
UPPER	04/07/90	1	2	3	4	5
UPPER	06/27/90	1	2	3	4	5

ORDER SPECIES	FAMILY	GENUS	TAXA	FG	LS
Amphipoda Gammarus sp.	Gammaridae	Gammarus	GAMM	CG	
Amphipoda Hyalella sp.	Talitridae	Hyalella	HYAL	CG	
Annelida Hirudinea	Hirudinea	Hirudinea	HIRU	PR	
Annelida Oligochaeta	Oligochaeta	Oligochaeta	OLIG	CG	
Coleoptera Dytiscidae sp.	Dytiscidae	Dytiscidae	DYTI	PR	L
Coleoptera Dubiraphia sp.	Elmidae	Dubiraphia	DUBA	CG	A
Coleoptera Optioservus sp.	Elmidae	Optioservus	OPTA	SC	A
Coleoptera Optioservus sp.	Elmidae	Optioservus	OPTI	SC	L
Coleoptera Stenelmis sp.	Elmidae	Stenelmis	STNA	SC	A
Coleoptera Stenelmis sp.	Elmidae	Stenelmis	STNL	SC	L
Coleoptera Gyrinidae	Gyrinidae	Gyrinidae	GYRI	PR	L
Coleoptera Berosus sp.	Hydrophilidae	Berosus	BERO	PH	L
Coleoptera Ectopria sp.	Psephenidae	Ectopria	ECTO	SC	L
Coleoptera Psephenus sp.	Psephenidae	Psephenus	PSEF	SC	L
Collembola Collembola	Collembola	Collembola	COLL	CG	
Decapoda Orconectes virilis	Cambaridae	Orconectes virilis	ORCO	SH	
Diptera Ceratopogonidae	Ceratopogonidae	Ceratopogonidae	CERA	PR	L
Diptera Ceratopogonidae	Ceratopogonidae	Ceratopogonidae	CERP	OT	P
Diptera Chironomidae	Chironomidae	Chironomidae	CHIP	OT	P
Diptera Chironomini	Chironomidae	Chironomini	CMNI	CG	L
Diptera Orthocladiinae	Chironomidae	Orthocladiinae	ORTH	CG	L
Diptera Tanypodinae	Chironomidae	Tanypodinae	TNYD	PR	L
Diptera Tanytarsini	Chironomidae	Tanytarsini	TNYT	CF	L
Diptera Culicidae	Culicidae	Culicidae	CULI	CF	L
Diptera Dixa sp.	Dixidae	Dixa	DIXA	CG	L

ORDER SPECIES	FAMILY	GENUS	TAXA	FG	LS
Diptera Chelifera sp.	Empididae	Chelifera	CHEL	PR	L
Diptera Hemerodromia sp.	Empididae	Hemerodromia	HEME	PR	L
Diptera Hemerodromia sp.	Empididae	Hemerodromia	HEMP	OT	P
Diptera Psychodidae	Psychodidae	Psychodidae	PSYC	CG	L
Diptera Simulium sp.	Simuliidae	Simulium	SIMI	CF	L
Diptera Simulium sp.	Simuliidae	Simulium	SIMP	OT	P
Diptera Stratiomyidae	Stratiomydae	Stratiomyidae	STRA	CG	L
Diptera Tabanidae	Tabanidae	Tabanidae	TABA	PR	L
Diptera Antocha sp.	Tipulidae	Antocha	ANTO	CG	L
Diptera Antocha sp.	Tipulidae	Antocha	ANTP	OT	P
Diptera Tipula sp.	Tipulidae	Tipula	TIPU	SH	L
Ephemeroptera Acentrella sp.	Baetidae	Acentrella	ACEN	CG	L
Ephemeroptera Baetis spp.	Baetidae	Baetis	BAET	CG	L
Ephemeroptera Caenis sp.	Caenidae	Caenis	CAEN	CG	L
Ephemeroptera Eurylophella aestiva	Ephemerellidae	Eurylophella	EPHE	CG	L
Ephemeroptera Stenacron interpunctatum	Heptageniidae	Stenacron	STIN	SC	L
Ephemeroptera Stenonema femoratum	Heptageniidae	Stenonema	STFE	SC	L
Ephemeroptera Paraleptophlebia sp.	Leptophlebiidae	Paraleptophlebia	PRLP	CG	L
Ephemeroptera Tricorythodes sp.	Tricorythidae	Tricorythodes	TRIC	CG	L
Gastropoda Anyclidae	Anyclidae	Anyclidae	ANCY	SC	
Gastropoda Physidae	Physidae	Physidae	PHYS	SC	
Gastropoda Planorbidae	Planorbidae	Planorbidae	PLAN	SC	
Hemiptera Corixidae	Corixidae	Corixidae	CORI	PH	
Hemiptera Gerris sp.	Gerridae	Gerris	GERR	PR	
Hemiptera Saldidae	Saldidae	Saldidae	SALD	PR	

ORDER SPECIES	FAMILY	GENUS	TAXA	FG	LS
Hemiptera Microvelia sp.	Veliidae	Microvelia	MICR	PR	
Hemiptera Paravelia sp.	Veliidae	Paravelia	PARA	PR	
Hemiptera Rhagovelia obesa	Veliidae	Rhagovelia	RHAG	PR	
Hydracarina Acarina	Acarina	Acarina	ACAR	PR	
Isopoda Caecidotea sp.	Asellidae	Caecidotea	ASEL	SH	
Isopoda Lirceus sp.	Asellidae	Lirceus	LIRC	SH	
Lepidoptera Petrophila sp.	Pyralidae	Petrophila	PETR	SC	L
Megaloptera Corydalus cornutus	Corydalidae	Corydalus	COCO	PR	L
Megaloptera Sialis sp.	Sialidae	Sialis	SIAL	PR	L
Nematoda Nematoda	Nematoda	Nematoda	NEMA	CG	
Odonata Calopterygidae	Calopterygidae	Calopterygidae	CALG	PR	L
Odonata Calopteryx sp.	Calopterygidae	Calopteryx	CALO	PR	L
Odonata Hetaerina sp.	Calopterygidae	Hetaerina	HETA	PR	L
Odonata Argia sp	Coenagrionidae	Argia	ARGI	PR	L
Pelecypoda Sphaeriidae	Sphaeriidae	Sphaeriidae	SPHA	CF	
Plecoptera Zealeuctra sp.	Leuctridae	Zealeuctra	LEUC	SH	L
Plecoptera Amphinemura sp.	Nemouridae	Amphinemura	AMPH	SH	L
Plecoptera Agnetina flavescens	Perlidae	Agnetina	AGFL	PR	L
Plecoptera Perlesta sp.	Perlidae	Perlesta	PERL	PR	L
Plecoptera Clioperla clio	Perlodidae	Clioperla	CLCL	PR	L
Trichoptera Cheumatopsyche sp.	Hydropsychidae	Cheumatopsyche	CHEP	OT	P
Trichoptera Cheumatopsyche sp.	Hydropsychidae	Cheumatopsyche	CHEU	CF	L
Trichoptera Hydropsyche sp.	Hydropsychidae	Hydropsyche	HYDP	OT	P
Trichoptera Hydropsyche sp.	Hydropsychidae	Hydropsyche	HYDR	CF	L
Trichoptera Hydroptila sp.	Hydroptilidae	Hydroptila	HPLP	OT	P

ORDER SPECIES	FAMILY	GENUS	TAXA	FG	LS
Trichoptera Hydroptila sp.	Hydroptilidae	Hydroptila	HPTL	PH	L
Trichoptera Hydroptilidae	Hydroptilidae	Hydroptilidae	HPLD	PH	L
Trichoptera Ochrotrichia sp.	Hydroptilidae	Ochrotrichia	OCHP	OT	P
Trichoptera Ochrotrichia sp.	Hydroptilidae	Ochrotrichia	OCHR	CG	L
Trichoptera Oxyethira sp.	Hydroptilidae	Oxyethira	OXYE	PH	L
Trichoptera Chimarra sp.	Philopotamidae	Chimarra	CHIM	CF	L
Trichoptera Polycentropus sp.	Polycentropodidae	Polycentropus	POLY	PR	L
Tricladida Dugesia sp.	Planariidae	Dugesia	DUGE	PR	

Bootstrapping BSTRAP program

The BSTRAP program allows the user to specify the name of the data file, the type of comparisons done, the classification level of organisms at which to do the analysis, and the data to be used for each community. Once these choices have been made, the program computes the following indices:

1) Diversity

   a) Simpson's D (Dominance)

   b) Shannon-Weiner H'

2) Similarity

   a) Pinkham and Pearson's B

   b) Jaccard's Index

For each index, a bootstrapping estimate of the standard error is computed, and confidence intervals are given using the bias-corrected percentile method. Please be aware that the nature of Jaccard's Index does not allow for accurate bootstrapping, since many of the bootstrap samples will have identical values. Thus, the estimate of the standard error, and the confidence intervals for this index should be used with caution.

The program also computes number of non-zero orders, families, genera, or taxon (depending on the level of analysis), the total number of individuals, and the EPT index for each community. The number of individuals in each non-zero order, family, genus, or taxa (depending on level of analysis) is printed for each community in the file FREQS.

For each of the similarity indices, Pinkham & Pearson's B and Jaccard's Index, a permutation based hypothesis test is performed on the null hypothesis of exactly equal communities (i.e., H_o: PPB = 1.00 and H_o: Jacc = .50). The p-value for each test is computed and printed, along with all other statistical results, in the file RESULTS.

Running BSTRAP

You must be in the NAPSAC subdirectory in dBase to run the BSTRAP program. At the prompt type

C:\DBASE\NAPSAC>**BSTRAP**

to start the bootstrapping program.

Required user responses.

1) Please enter the name of the data file:

   If file not found, a run-time error occurs, and program is terminated.

2) Do you want:

   a) Diversity Indices for a single community, or
   b) Diversity and Similarity Indices for two communities (Type 1 or 2, then enter)?

3) At what level of Taxa do you want to make comparisons?

   a) Order
   b) Family
   c) Genus
   d) Taxa

   Enter level and press enter.

4) Please enter the No. of site, date combinations in community 1.

NOTE: Community 1 can consist of one or more samples (site, date combinations). For example community 1 can be comprised of one site, date combination or if you wish to compare one year to another at a single site, community 1 may be a combination of 3 or 4 site, date combinations if the site had been sampled in the spring, summer, fall and/or winter.

5) Enter SITE for combination 1. It MUST BE UP TO a 4 letter code, in **CAPS**. (Repeated as many times as entered on 4)

6) Enter DATE for combination 1. It MUST BE an 8 number code (YYYYMMDD). (Repeated as many times as entered on 4)

7) You entered Sites, Dates. Is this correct? (0 = no, 1 = yes)

8-11) Same as #4-7, but for community 2.

The number of individuals in each non-zero order, family, genus, or taxa (depending on level of analysis) is printed for each community in the file FREQS. The p-value for each test is computed and printed, along with all other statistical results, in the file RESULTS.

Both the FREQS and RESULTS files are in ASCII format can be typed on the screen or sent to a printer using DOS commands (TYPE or PRINT) or imported into various word processing programs for editing. Again if community analysis is repeated these files will be over written and output lost. Before further analysis is performed output files can be renamed to prevent being over written.

PROGRAM OUTPUTS

To generate examples of various outputs from the NAPSAC and BSTRAP programs, two sites collected on 4 dates each with 5 replicates were used. First, the Taxa Codes, Orders, Families, Genus, Species, Life stages, and Functional Groups were entered into the Taxa Dictionary file. Then selected '2' from the Main menu and the Taxa Dictionary Maintenance menu is displayed.

```
                    National Parks NAPSAC System
                    Taxa Dict Maintenance Screen

ADD Mode

Taxa: SIMU        Order: DIPTERA__________
                 Family: SIMULIIDAE_______
                  Genus: SIMULIUM_________
                Species: SIMULIUM SPP.___________________________
             Life Stage: L

Enter a "9" in TAXA to Exit without updating the file

                          Messages:
```

This process is repeated until all taxa information is entered into the Dictionary. Entering a '9' in the Taxa field to exit the current function and return to the Taxa Dictionary Maintenance menu.

Sample records are added by choosing "1" from the main menu. When entering Sample data, the operator will enter the Site, Date, Replicate, Taxa Code, and number of occurrences of the Taxa on the following screen:

```
National Parks NAPSAC System
 Sample Maintenance Screen

_______ Mode
                                                           Number
Site: ____  Date: __/__/__ Replicate: __  Taxa Code: ___   Found: ____

Enter a "9" in SITE to Exit without updating the file

     Order: ____________________          Family: ____________________
     Genus: ____________________
   Species: ________________________________________
Life Stage: _                     Funct Group: __

                        Messages:
```

When the screen is displayed, all the fields except Mode will say Add. When the Taxa Code field is filled, the program will read the Taxa Dictionary file to retrieve the Order, Family, etc. and display that information on the screen. The Message area is used to send messages to the operator. Enter a '9' in the Site field to exit and return to the Sample Maintenance menu when all samples have been entered.

When "3" is selected from the Main menu, the Report menu will be displayed:

```
─────────── NATIONAL PARKS NAPSAC - SAMPLE COLLECTION SYSTEM ───────────

    STANDARD REPORTS              FILE DUMPS

A- SITE/DATE/REP LIST          J- SAMPLE          P- OUTPUT ASCII FILE
B-                             K- TAXA DICT       R-
C-                             L-                 S-
D-                             M-                 T-
E- VALIDATION ERROR REPORT     N-
F-                             O-
G-
H-
I-

              ENTER "Q" TO RETURN TO THE MAIN MENU

              PLEASE ENTER YOUR CHOICE: _
```

To get a report of all the Site, Date, Replicate combinations in the Sample file choose "A". The following report will be generated:

```
08/15/91          LIST OF SITE, DATE, & REPS, IN THE SAMPLE FILE

SITE    DATE     REPLICATES
----- -------- ------------------------------------------------
CAR   06/27/89  1  2  3  4  5
CAR   09/13/89  1  2  3  4  5
CAR   12/28/89  1  2  3  4  5
CAR   04/20/90  1  2  3  4  5
WIL   06/27/89  1  2  3  4  5
WIL   09/13/89  1  2  3  4  5
WIL   12/28/89  1  2  3  4  5
WIL   04/20/90  1  2  3  4  5
```

The SAMPLE FILE report is a list of all or some of the records in the Sample file. It lists the Site, Date, Rep, Taxa, and number of occurrences. If you want all the records on file, just press return in the Site, Date, and Replicate field. If you want all the records for a particular Site, enter the Site and leave the other fields blank. For example, if you want to check if the first replicate for "CAR" collected on 06/27/89 was entered correctly you would want to type "CAR" in as the Site, "06/27/89" in for the Date, and "1" in the Replicate field. The following report will be generated:

```
 08/15/91                                NAPSAC SYSTEM
                                      SAMPLE DATA LISTING
SITE     DATE       REP  TAXA   OCCUR
-----    --------   ---  ----   -----
CAR      06/27/89    4   DUGE     17
CAR      06/27/89    4   HAAM      7
CAR      06/27/89    4   MIBU      1
CAR      06/27/89    4   NEPP      4
CAR      06/27/89    4   OLIG      5
CAR      06/27/89    4   OPTI      1
CAR      06/27/89    4   ORTH     19
CAR      06/27/89    4   PECI      1
CAR      06/27/89    4   PLEU      2
CAR      06/27/89    4   POCI      3
CAR      06/27/89    4   PSHE      6
CAR      06/27/89    4   SIMU      7
CAR      06/27/89    4   STAL     10
CAR      06/27/89    4   STEA      1
CAR      06/27/89    4   STEN      5
CAR      06/27/89    4   TABA      1
CAR      06/27/89    4   TIPU      1
CAR      06/27/89    4   TNYP      8
CAR      06/27/89    4   TNYT      8
CAR      06/27/89    4   ZECL     74
CAR      06/27/89    5   ARPL      3
CAR      06/27/89    5   BAET      1
CAR      06/27/89    5   CHAT      1
CAR      06/27/89    5   CHEU      3
CAR      06/27/89    5   DUGE      1
CAR      06/27/89    5   GERR      1
CAR      06/27/89    5   HAAM      2
CAR      06/27/89    5   HEME      1
CAR      06/27/89    5   NEPP      5
CAR      06/27/89    5   ORTH      4
CAR      06/27/89    5   POCI      2
CAR      06/27/89    5   PSHE      2
CAR      06/27/89    5   TNYT      2
CAR      06/27/89    5   ZECL     12
```

If you leave the replicate field blank all of the replicates will be reported:

08/15/91 NAPSAC SYSTEM PAGE: 1

SAMPLE DATA LISTING

SITE	DATE	REP	TAXA	OCCUR
CAR	06/27/89	1	ACAR	5
CAR	06/27/89	1	ARPL	5
CAR	06/27/89	1	BAET	2
CAR	06/27/89	1	CHAT	3
CAR	06/27/89	1	CHEU	14
CAR	06/27/89	1	CHIP	1
CAR	06/27/89	1	DIXA	1
CAR	06/27/89	1	DUGE	6
CAR	06/27/89	1	HAAM	10
CAR	06/27/89	1	MIBU	1
CAR	06/27/89	1	NEPP	14
CAR	06/27/89	1	OLIG	1
CAR	06/27/89	1	OPTA	2
CAR	06/27/89	1	ORTH	3
CAR	06/27/89	1	POCI	1
CAR	06/27/89	1	PSHA	2
CAR	06/27/89	1	PSHE	8
CAR	06/27/89	1	STAL	2
CAR	06/27/89	1	STEA	6
CAR	06/27/89	1	TNYP	3
CAR	06/27/89	1	TNYT	16
CAR	06/27/89	1	ZECL	31
CAR	06/27/89	2	ACAR	1
CAR	06/27/89	2	CHEU	3
CAR	06/27/89	2	DECA	1
CAR	06/27/89	2	HAAM	1
CAR	06/27/89	2	NEPP	27
CAR	06/27/89	2	PLEU	2
CAR	06/27/89	2	POCI	3
CAR	06/27/89	2	STEA	1
CAR	06/27/89	2	TNYT	7
CAR	06/27/89	2	ZECL	5
CAR	06/27/89	3	ARPL	3
CAR	06/27/89	3	BAET	1
CAR	06/27/89	3	CHAT	4
CAR	06/27/89	3	CHEU	12
CAR	06/27/89	3	HAAM	1
CAR	06/27/89	3	MIBU	2
CAR	06/27/89	3	NEPP	21
CAR	06/27/89	3	ORTH	1
CAR	06/27/89	3	PLEU	2
CAR	06/27/89	3	POCI	2
CAR	06/27/89	3	PSHE	3
CAR	06/27/89	3	STAL	2
CAR	06/27/89	3	STEA	1
CAR	06/27/89	3	TNYP	3
CAR	06/27/89	3	TNYT	1
CAR	06/27/89	3	ZECL	7
CAR	06/27/89	4	ARPL	63
CAR	06/27/89	4	CHAT	26
CAR	06/27/89	4	CHEU	19
CAR	06/27/89	4	CHIP	1
CAR	06/27/89	4	DUGE	17
CAR	06/27/89	4	HAAM	7
CAR	06/27/89	4	MIBU	1
CAR	06/27/89	4	NEPP	4
CAR	06/27/89	4	OLIG	5
CAR	06/27/89	4	OPTI	1
CAR	06/27/89	4	ORTH	19
CAR	06/27/89	4	PECI	1
CAR	06/27/89	4	PLEU	2
CAR	06/27/89	4	POCI	3
CAR	06/27/89	4	PSHE	6
CAR	06/27/89	4	SIMU	7
CAR	06/27/89	4	STAL	10
CAR	06/27/89	4	STEA	1
CAR	06/27/89	4	STEN	5
CAR	06/27/89	4	TABA	1
CAR	06/27/89	4	TIPU	1
CAR	06/27/89	4	TNYP	8
CAR	06/27/89	4	TNYT	8
CAR	06/27/89	4	ZECL	74
CAR	06/27/89	5	ARPL	3
CAR	06/27/89	5	BAET	1
CAR	06/27/89	5	CHAT	1
CAR	06/27/89	5	CHEU	3
CAR	06/27/89	5	DUGE	1
CAR	06/27/89	5	GERR	1
CAR	06/27/89	5	HAAM	2
CAR	06/27/89	5	HEME	1
CAR	06/27/89	5	NEPP	5
CAR	06/27/89	5	ORTH	4
CAR	06/27/89	5	POCI	2
CAR	06/27/89	5	PSHE	2
CAR	06/27/89	5	TNYT	2
CAR	06/27/89	5	ZECL	12

If you want all the records for a particular Site on a particular date, enter the Site and Date and leave Replicate blank. Any combination of Site, Date, and Replicate can be entered.

If "K" is chosen a list of all the records in the Taxa Dictionary is generated. It lists the Taxa Code, Order, Family, Genus, Species, Life Stage, and Functional group. The following is an example of the Taxa Dictionary:

```
08/15/91                        NAPSAC SYSTEM                       PAGE:     1
                         TAXA DICTIONARY FILE LISTING

ORDER                 FAMILY                GENUS                 TAXA  FG  LS
  SPECIES
--------------------  --------------------  --------------------  ----  --  --
Amphipoda             Amphipoda             Amphipoda             AMPH  CG
  Amphipoda
Annelida              Hirudinea             Hirudinea             HIRU  PR
  Hirudinea
Annelida              Oligochaeta           Oligochaeta           OLIG  CG
  Oligochaeta
Coleoptera            Elmidae               Dubiraphia            DUBA  CG  A
  Dubiraphia sp.
Coleoptera            Elmidae               Dubiraphia            DUBI  CG  L
  Dubiraphia sp.
Coleoptera            Elmidae               Optioservus           OPTA  SC  A
  Optioservus sp.
Coleoptera            Elmidae               Optioservus           OPTI  SC  L
  Optioservus sp.
Coleoptera            Elmidae               Stenelmis             STEA  SC  A
  Stenelmis spp.
Coleoptera            Elmidae               Stenelmis             STEN  SC  L
  Stenelmis spp.
Coleoptera            Psephenidae           Psephenus             PSHA  SC  A
  Psephenus herricki
Coleoptera            Psephenidae           Psephenus             PSHE  SC  L
  Psephenus herricki
Collembola            Collembola            Collembola            COLL  CG
  Collembola
Decapoda              Cambaridae            Cambaridae            DECA  SH
  Cambaridae
Diptera               Ceratopogonidae       Ceratopogonidae       CERA  PR  L
  Ceratopogonidae
Diptera               Chironomidae          Chironomidae          CHIP  OT  P
  Chironomidae
Diptera               Chironomidae          Chironomini           CNMI  CG  L
  Chironomini
Diptera               Chironomidae          Diamesinae            DIAM  CG  L
  Diamesinae
Diptera               Chironomidae          Orthocladiinae        ORTH  CG  L
  Orthocladiinae
Diptera               Chironomidae          Tanypodinae           TNYP  PR  L
  Tanypodinae
Diptera               Chironomidae          Tanytarsini           TNYT  CF  L
  Tanytarsini
Diptera               Dixidae               Dixa                  DIXA  CG  L
  Dixa sp.
Diptera               Empididae             Hemerodromia          HEME  PR  L
  Hemerodromia sp.
Diptera               Empididae             Hemerodromia          HEMP  OT  P
  Hemerodromia sp.
Diptera               Simuliidae            Simulium              SIMP  OT  P
  Simulium spp.
Diptera               Simuliidae            Simulium              SIMU  CF  L
  Simulium spp.
```

ORDER SPECIES	FAMILY	GENUS	TAXA	FG	LS
Diptera Tabanidae	Tabanidae	Tabanidae	TABA	PR	L
Diptera Antocha sp.	Tipulidae	Antocha	ANTO	CG	L
Diptera Tipula sp.	Tipulidae	Tipula	TIPU	SH	L
Ephemeroptera Acentrella insignificans	Baetidae	Acentrella	ACEN	SC	L
Ephemeroptera Diphetor hageni	Baetidae	Diphetor	BAET	CG	L
Ephemeroptera Caenis sp.	Caenidae	Caenis	CAEN	CG	L
Ephemeroptera Stenacron interpunctatum	Heptageniidae	Stenacron	STIN	SC	L
Ephemeroptera Stenonema femoratum	Heptageniidae	Stenonema	STFE	SC	L
Ephemeroptera Habrophlebiodes americana	Leptophlebiidae	Habrophlebiodes	HAAM	SC	L
Gastropoda Ancylidae	Ancylidae	Ancylidae	ANCY	SC	
Gastropoda Hydrobiidae	Hydrobiidae	Hydrobiidae	HYBI	SC	
Gastropoda Planorbidae	Planorbidae	Planorbidae	PLNO	SC	
Gastropoda Pleuroceridae	Pleuroceridae	Pleuroceridae	PLEU	SC	
Hemiptera Corixidae	Corixidae	Corixidae	CORI	PH	L
Hemiptera Gerris remigis	Gerridae	Gerris	GERR	PR	L
Hemiptera Trepobates sp.	Gerridae	Trepobates	TRPO	PR	L
Hemiptera Microvelia buenoi	Veliidae	Microvelia	MIBU	PR	L
Hemiptera Rhagovelia obesa	Veliidae	Rhagovelia	RHAG	PR	L
Hydracarina Hydracarina	Hydracarina	Hydracarina	ACAR	PR	
Isopoda Isopoda	Isopoda	Isopoda	ISPD	SH	
Lepidoptera Petrophila sp.	Pyralidae	Petrophila	PETR	SC	L
Megaloptera Corydalus cornutus	Corydalidae	Corydalus	COCO	PR	L
Megaloptera Sialis infumata	Sialidae	Sialis	SIAL	PR	L
Odonata Argia plana	Coenagrionidae	Argia	ARPL	PR	L
Odonata Stylogomphus albistostylus	Gomphidae	Stylogomphus	STAL	PR	L
Trichoptera Hydropsyche sp.	Hydropsychidae	Hydropsyche	HYDR	CF	L
Trichoptera Ocotrichia sp.	Hydroptilidae	Ocotrichia	OCHR	CG	L
Trichoptera Oxyethira dualis	Hydroptilidae	Oxyethira	OXYE	PH	L
Trichoptera Oxyethira dualis	Hydroptilidae	Oxyethira	OXYP	OT	P
Trichoptera Neophylax fuscus	Limnephilidae	Neophylax	NEPH	SC	L
Trichoptera Neophylax fuscus	Limnephilidae	Neophylax	NEPP	OT	P
Trichoptera Pycnopsyche sp.	Limnephilidae	Pycnopsyche	PYCN	SH	L
Trichoptera Pycnopsyche sp.	Limnephilidae	Pycnopsyche	PYCP	OT	P
Trichoptera Chimarra atterrima	Philopotamidae	Chimarra	CHAP	OT	P
Trichoptera Chimarra atterrima	Philopotamidae	Chimarra	CHAT	CF	L
Trichoptera Polycentropus cinerus	Polycentropodidae	Polycentropus	POCI	PR	L
Trichoptera Lype diversa	Psychomyiidae	Lype	LYDI	SC	L

If "P" is chosen the program will create an ASCII file (**SAMPOUT.ASC**) that can be loaded into the bootstrapping BSTRAP system. When the operator runs this program, it will ask for the Beginning and Ending dates of the samples to be included in the output.

After creating your output ASCII file, return to the main menu and exit the NAPSAC program which will then return you to the NAPSAC subdirectory within dBase. To start the bootstrapping program type **BSTRAP**. The program will ask you to:

1) Please enter the name of the data file:

type **SAMPOUT.ASC**

2) Do you want:

a) Diversity Indices for a single community, or
b) Diversity and Similarity Indices for two communities (Type 1 or 2, then enter)?

For example, if "**2**" is chosen the program will ask you

3) At what level of Taxa do you want to make comparisons?

a) Order
b) Family
c) Genus
d) Taxa

Enter level and press enter.

for this example the response chosen was "**d**"

4) Please enter the No. of site, date combinations in community 1.

In this example we will compare the sample (comprised of the 5 replicates) from the "CAR" site taken on 06/27/89 to the sample from the "WIL" site collected on the sample date.

Enter "**1**".

However, community 1 can consist of one or more samples (site, date combinations). For example community 1 can be comprised of one site, date combination or if you wish to compare one year to another at a single site, community 1 may be a combination of 3 or 4 site, date combinations if the site had been sampled in the spring, summer, fall and/or winter.

5) Enter SITE for combination 1. It MUST BE UP TO a 4 letter code, in **CAPS**. (Repeated as many times as entered on 4)

Enter **CAR**

6) Enter DATE for combination 1. It MUST BE an 8 number code (YYYYMMDD). (Repeated as many times as entered on 4)

Enter **19890627**

7) You entered Sites, Dates. Is this correct? (0 = no, 1 = yes)

CAR 19890627

enter **1** to indicated that this is correct.

At this time the computer will generate diversity for community 1.

When this is complete it will ask you for the same information about concerning community 2.

8) Please enter the No. of site, date combinations in community 2.

Again enter **1**

9) Enter SITE for combination 1. It MUST BE UP TO a 4 letter code, in **CAPS**. (Repeated as many times as entered on 4)

Enter **WIL**

6) Enter DATE for combination 1. It MUST BE an 8 number code (YYYYMMDD). (Repeated as many times as entered on 4)

Enter **19890627**

7) You entered <u>Sites</u>, <u>Dates</u>. Is this correct? (0 = no, 1 = yes)

WILL 19890627

enter 1 to indicate that this is correct.

At this time the computer will generate diversity for community 2 and similarity indices for the two communities.

When the program is complete the number of individuals in each non-zero order, family, genus, or taxa (depending on level of analysis) is printed for each community in the file FREQS. The p-value for each test is computed and printed, along with all other statistical results in the file RESULTS. Both the FREQS and RESULTS files are in ASCII format and can be typed on the screen or sent to a printer using DOS commands (TYPE or PRINT), or imported into various word processing programs for editing.

The following is the out put from the RESULTS file created from the example:

```
BOOTSTRAP RESULTS
*****************************************************************
*****************************************************************
      DATA SET IS              sampout.asc
      ANALYZED AT LEVEL        TAXA
*****************************************************************
*****************************************************************

      RESULTS FOR COMMUNITY 1:    SITES     DATES
                           CAR    19890627

               NUMBER OF NON-ZERO TAXON    =  32
               TOTAL NUMBER OF INDIVIDUALS =   584.

EPT =   8

  ----------------------------------------------------------
        SIMPSONS D =    .10110
```

MEAN OF INDEX OVER ALL BOOTSTRAP SAMPLES IS .11009
AND THE STANDARD ERROR OF THE MEAN IS .01751

CONFIDENCE INTERVALS OBTAINED USING THE
BIAS CORRECTED PERCENTILE METHOD ARE

	LIMITS	
LEVEL	(LOWER,	UPPER)
80	(.09622,	.10728)
90	(.09622,	.11131)
95	(.09622,	.11569)

SHANNON-WEINER H = 2.66658

MEAN OF INDEX OVER ALL BOOTSTRAP SAMPLES IS 2.59838
AND THE STANDARD ERROR OF THE MEAN IS .09857

CONFIDENCE INTERVALS OBTAINED USING THE
BIAS CORRECTED PERCENTILE METHOD ARE

	LIMITS	
LEVEL	(LOWER,	UPPER)
80	(2.64878,	2.69822)
90	(2.62322,	2.69822)
95	(2.60090,	2.69822)

RESULTS FOR COMMUNITY 2: SITES DATES
WIL 19890627

NUMBER OF NON-ZERO TAXON = 37
TOTAL NUMBER OF INDIVIDUALS = 2375.

EPT = 12

SIMPSONS D = .10118

MEAN OF INDEX OVER ALL BOOTSTRAP SAMPLES IS .10890
AND THE STANDARD ERROR OF THE MEAN IS .00811

CONFIDENCE INTERVALS OBTAINED USING THE
BIAS CORRECTED PERCENTILE METHOD ARE

```
              LIMITS
   LEVEL  (  LOWER,   UPPER)
     80   (  .09618,  .10397)
     NOTE---LOWER LIMIT IS LESS THAN THE MINIMUM
      VALUE, SO THE MINIMUM VALUE WAS USED BELOW
     90   (  .09618,  .10625)
     NOTE---LOWER LIMIT IS LESS THAN THE MINIMUM
      VALUE, SO THE MINIMUM VALUE WAS USED BELOW
     95   (  .09618,  .11010)
```

SHANNON-WEINER H = 2.51730

MEAN OF INDEX OVER ALL BOOTSTRAP SAMPLES IS 2.48378
AND THE STANDARD ERROR OF THE MEAN IS .04716

CONFIDENCE INTERVALS OBTAINED USING THE
BIAS CORRECTED PERCENTILE METHOD ARE

```
              LIMITS
   LEVEL  (  LOWER,   UPPER)
     80   ( 2.48578, 2.57683)
     90   ( 2.47492, 2.57683)
     95   ( 2.46101, 2.57683)
```

SIMILARITY INDICES BETWEEN COMMUNITY 1 & COMMUNITY 2

PINKHAM & PEARSONS B = .11212
NUMBER OF NON-ZERO TAXON = 49

MEAN OF INDEX OVER ALL BOOTSTRAP SAMPLES IS .12742
AND THE STANDARD ERROR OF THE MEAN IS .02224

CONFIDENCE INTERVALS OBTAINED USING THE
BIAS CORRECTED PERCENTILE METHOD ARE

	LIMITS	
LEVEL	(LOWER,	UPPER)
80	(.07576,	.12762)
90	(.06273,	.13679)
95	(.05903,	.14441)

JACCARDS INDEX = .28986
NUMBER OF NON-ZERO TAXON = 49

MEAN OF INDEX OVER ALL BOOTSTRAP SAMPLES IS .27812
AND THE STANDARD ERROR OF THE MEAN IS .01608

CONFIDENCE INTERVALS OBTAINED USING THE
BIAS CORRECTED PERCENTILE METHOD ARE

	LIMITS	
LEVEL	(LOWER,	UPPER)
80	(.28125,	.30769)
90	(.27586,	.30909)
95	(.26984,	.31250)

FOR COMMUNITY 1 AND COMMUNITY 2
RESULTS OF THE PERMUTATION TEST FOR THE HYPOTHESIS
OF COMPLETELY SIMILAR COMMUNITIES.

EXACT PERMUTATION TEST FOR SIMILARITY:

THE VALUE OF PINKHAM & PEARSONS B = .11212
THE P-VALUE FOR THIS INDEX IS P = .0000

THE VALUE OF JACCARDS INDEX = .28986
THE P-VALUE FOR THIS INDEX IS P = .0000

The following is the FREQS file created in the previous example:

BOOTSTRAP **FREQS**

```
GROUP 1               SITES    DATES
                        CAR      19890627


TAXA                  FREQUENCY   TAXA              FREQUENCY


OLIG                        6.    OPTA                     2.
OPTI                        1.    STEA                     9.
STEN                        5.    PSHA                     2.
PSHE                       19.    DECA                     1.
CHIP                        2.    ORTH                    27.
TNYP                       14.    TNYT                    34.
DIXA                        1.    HEME                     1.
SIMU                        7.    TABA                     1.
TIPU                        1.    BAET                     4.
HAAM                       21.    PLEU                     6.
GERR                        1.    MIBU                     4.
ACAR                        6.    ARPL                    74.
STAL                       14.    ZECL                   129.
PECI                        1.    CHEU                    51.
NEPP                       71.    CHAT                    34.
POCI                       11.    DUGE                    24.


GROUP 2               SITES    DATES
                        WIL      19890627


TAXA                  FREQUENCY   TAXA              FREQUENCY
```

AMPH	372.	HIRU	4.
OLIG	82.	DUBI	1.
OPTA	5.	OPTI	116.
STEA	22.	STEN	286.
DECA	5.	CHIP	25.
CNMI	206.	TNYP	37.
TNYT	242.	HEME	5.
SIMP	1.	TIPU	1.
BAET	1.	CAEN	1.
STIN	6.	PLEU	3.
TRPO	1.	MIBU	2.
RHAG	2.	ISPD	174.
COCO	1.	SIAL	6.
ARPL	6.	ZECL	11.
CHEP	1.	CHEU	115.
OCHR	1.	OXYP	3.
NEPP	36.	PYCN	1.
CHAP	6.	CHAT	220.
DUGE	368.		

CLASSIFICATION OF THREATS

CLASSIFICATION OF THREATS

	PHYSICAL							CHEMICAL															BIOLOGICAL		
								Nutrients						Toxicants											
ACTIVITY	sediment	turbidity	thermal	water level	discharge			nitrogen	phosphorus	micro-nutrients	organics			heavy metals	ammonia	chlorine	acid	pesticides	industrial organics	hydrocarbons			exotic species	disease	exploitation
1. Industry																									
paper, pulp	4																								
mining, refining	4																								
electronics		2		2	2									2					1						
textile																									
food processing	2	2	2	2	2									2	2	2									
2. Energy																									
coal mining	4																								
thermal power plants				2	2																				
hydropower stations	4																								
power lines	4																								
fuel storage																									
domestic heating	4																								

1 = Known heavy, 2 = Known light, 3 = Expected in future, 4 = absent

	PHYSICAL							CHEMICAL															BIOLOGICAL		
								Nutrients						Toxicants											
ACTIVITY	sediment	turbidity	thermal	water level	discharge			nitrogen	phosphorus	micro-nutrients	organics			heavy metals	ammonia	chlorine	acid	pesticides	industrial organics	hydrocarbons			exotic species	disease	exploitation
3. Transport																									
railroads		2		2	2															1					
roads				2																2					
boats	4																								
4. Agriculture																									
overgrazing	4																								
fertilizers								2	2		2														
management chemicals																									
5. Forestry																									
woodcutting	4																								
transport	4																								
processing	4																								
fire regimes	4																								

1 = Known heavy, 2 = Known light, 3 = Expected in future, 4 = absent

ACTIVITY	PHYSICAL							CHEMICAL															BIOLOGICAL		
								Nutrients						Toxicants											
	sediment	turbidity	thermal	water level	discharge			nitrogen	phosphorus	micro-nutrients	organics			heavy metals	ammonia	chlorine	acid	pesticides	industrial organics	hydrocarbons			exotic species	disease	exploitation
6. Fisheries																									
commercial	4																								
recreational	4																								
7. Urbanization																									
sewage	1		2	1	1			2	2	2	1			1				2	2						
solid waste																									
runoff				1				2	2	2	2							1		2					
infra-structure																									
8. Recreation																									
resort development	4																								
camping	4																								
boating	4																								
poaching	4																								
fishing	4																								

1 = Known heavy, 2 = Known light, 3 = Expected in future, 4 = absent

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ADDITIONAL READINGS

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North American prairie streams as systems for ecological study*

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Abstract. The Great Plains and Osage Plains of interior North America included vast prairie regions before settlement by western man. Prairie streams that exist today are ecologically interesting for their unstable flow regimes and harsh fluctuations in environmental conditions. How much extant prairie streams have changed from their pre-settlement conditions is unknown, but writings of early explorers suggest that then, as now, mainstream flows were highly variable. Although historically, some prairie rivers were turbid, the siltiness of others has increased with agricultural expansions, and some mainstream biota (fishes) are known to have disappeared. Small streams in these regions have probably changed even more; although they were clear 100 years ago, many are now highly turbid. Differences in ecological conditions among streams of the prairie region may be as great as those between prairie and nearby upland streams. For example, streams of the southern plains are characterized by irregular flow and substrates of small particle size, whereas streams of the northern prairies are more consistent in flow and many have cobble substrates. For some prairie streams, published studies exist of many ecological features, such as hydrologic regimes, productivity, respiration, organic matter processing, and composition of the biota; however, such basic features are unknown for most streams of the interior plains. Limited measurements suggest that typical central or southern prairie streams differ from streams of nearby uplands or northern forested regions in functional properties and biotic composition. Community structure and ecological functioning of many prairie streams appear strongly influenced by physicochemical limitations resulting from irregular flow regimes and environmental harshness, such as major disturbances due to drought and flood. Despite the above conditions, numerous examples suggest that prairie stream faunas are also influenced by biotic interactions, including those due to multi-trophic level effects. A relatively new approach in stream ecology applies the analysis of complex hydraulic parameters to questions concerning distributions or adaptations of the biota. This approach may be profitable in prairie streams, but needs modification to include the sometimes lengthy periods of low or no flow in these systems. As a result of the discussions giving rise to this paper, numerous specific topics are suggested for future investigations. These can be generalized in four categories: (1) basic description and comparison of biotas, processes, or rates; (2) biotic adaptations; (3) controlling mechanisms; and (4) comparisons of prairie stream ecosystems with those of other kinds of temperate streams in North America. Streams of the Mississippi Embayment, Interior Highlands, the Rocky and Appalachian mountains, and the prairies of our inland plains provide a contrasting array of study sites at a similar latitude for consideration by ecologists desiring a broad comparative or experimental approach to questions in stream ecology.

Key words: prairie streams, historical changes, hydrologic regime, productivity, microbial processing, organic matter dynamics, biotic interactions, disturbance, physicochemical limitations, stream biota.

The region

The Great Plains and the Osage Plains, a vast expanse of relatively flat land reaching from the Ozark and Ouachita uplands to the foothills of the Rocky Mountains (Fig. 1), included the widest reach of unbroken prairie that existed in North America two centuries ago. Geologically, the Osage Plains are part of the Central Lowland which covers about 1,680,000 km² in the central United States and Canada (Hunt 1974). The Osage Plains rise toward the west to an altitude of approximately 600 m at the 100th meridian, which is an approximate boundary between the Central Lowland and the Great Plains Physiographic Province. The Great Plains Province, including the Alberta Plain of Canada

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FIG. 1. Location of the Great Plains and the Osage Plains of North America, and of the Interior Highlands (Ozark and Ouachita uplands).

(Fig. 1), includes approximately 1,490,000 km^2 of semiarid land (Hunt 1974), rising from the east to an altitude of about 1675 m at the base of the Rocky Mountains. While there are details of difference in structural geology of the Great Plains and the Central Lowland, Hunt (1974) showed that these two formations are separated mostly by the 100th meridian, the 600-m contour, the 50-cm rainfall line, a boundary between tall and short grasses, and the eastern limit of Tertiary formations that contain sediments eroded from the Rocky Mountains and washed onto the plains.

These interior plains are not all homogeneous geologically or biologically. For example, in many places they are not flat. The Flint Hills of eastern Kansas have relatively sharp relief, with flat caprock mesas grading downward through grassy slopes to cottonwood-lined streams similar to gravel streams of the Ozark upland. In Oklahoma, the Arbuckle and Wichita Mountains contrast sharply with the prairie, and generally the topography is more undulating than the words "plain" or "prairie" suggest. Large salt flat or sand dune areas exist, and the Black Mesa in the Oklahoma panhandle provides sharp vertical relief in sandstone. These local phenomena aside, however, the interior plains are united collectively by relatively low relief, highly variable rainfall, and, with the exception of stream borders, more open grassland than forest. The forested reaches that do exist, like the "crosstimbers" of southern Kansas, central Oklahoma, and north Texas, consist of trees that seem dwarfed compared with their eastern counterparts.

The central plains region is dissected by major rivers like the Missouri, Platte, Kansas, Arkansas, Cimarron, Canadian, Washita, and Red that drain generally from west to east (Fig. 2). The larger of these systems have headwaters in the eastern slopes of the Rocky Mountains or the high plains of eastern Colorado or North Texas, and roughly parallel one another en route to the Mississippi River or the Gulf of Mexico. The southern Great Plains is drained by streams of the Texas gulf coast, including the Trinity, Brazos, Colorado, and Pecos Rivers (Fig. 2). In Canada, the upper Great Plains are drained by the vast McKenzie, Peace, and Saskatchewan River systems.

It is relatively easy with standard references to delineate geographically this interior plains or prairie region and the major drainages therein. Much more difficult is the task of defining a "prairie stream" as a unique type that can be discussed in the context of modern aquatic ecology. For example, cool and warm water rivers of the northern Great Plains may have higher gradient, coarser substrates, and more persistent flow than similar-sized streams farther south in Kansas, Oklahoma, or Texas (J. A. Gore, personal communication). Because of the difficulty of generalizing about prairie streams from Canada to the southern United States, much of this paper focuses upon streams of the central and southern plains with which I am more familiar, and which were the subject of most discussion in the Prairie Streams Workshop.

Existing conditions

Another difficulty is knowing whether or not the prairie streams that we perceive today are reasonable facsimiles of those a traveler would have found 200 years ago, or if they are mere remnants of the former systems, having been ravaged by pump, plow, and pollution. Prairie streams of the vast interior plains range in size from the large river mainstreams to tens of thousands of kilometers of intermittent or ephemeral streams that flow only during part of the year. A few years ago, I collected minnows in streams of west Kansas, taking large numbers of fish in the Smoky Hill and other small rivers in the area. Shortly thereafter I attempted another collecting trip on a north to south axis some 40 kilometers west of my previous route. The trip was a loss, as the same streams slightly farther west had very little or no water. Even the mainstreams can be elusive. I have collected fish in the Arkansas River at Great Bend, Kansas, where the water was knee to waist deep, 50 m or so wide, and with substantial flow although there had been no recent rains. By traveling upstream a few kilometers, I have stood with one foot on each bank of this same Arkansas River mainstream, where there was scarcely enough water for mosquitofish and a few red shiner minnows. In a similar fashion, other southwestern rivers such as the Rio Grande and the Canadian wax and wane through their course downstream, responding not only to vagaries of rainfall but to withdrawal of irrigation or municipal water from the river channels or porous aquifers.

Despite tremendous variation among mainstreams of the prairies, tributary creeks can be even more unpredictable. In many prairie creeks "you haven't been there until you've been there a whole year", and this probably should be stated as "five years" or "ten years" or more. Uncertainty among days, seasons, and years seems to be normal rather than unusual in these streams. "Stream" research begun in boldly flowing streams in southern Oklahoma can be frustrated in late summer as flow ceases and moisture disappears from desiccated streambeds. In Marshall County, Oklahoma, numerous creeks seem fairly similar most of the year: steep soil-gravel banks, beds of sand and gravel over coarse fragmented limestone, and relatively clear water with substantial flow. However, in the entire county only Brier Creek, with its numerous seep springs, flows in late summer (Power and Matthews 1983) while all other creeks lose surface flow and may dry completely.

Despite the wide variations in streams of the central and southern prairies, the region has some generally definable stream types, from largest mainstreams to smallest creeks. All have in common a potential lack of water. Beyond limitations in amounts of water, prairie streams seem most differentiated by geology, which determines the substratum over which they flow.

FIG. 2. Major drainages of the interior plains of the United States.

Many prairie mainstreams, defying the popular concept that they are muddy, are actually rather clear except immediately after rains. Rivers flowing over sand, like the Ninnesecah or the Chikaskia Rivers of southern Kansas, can be very clear, and even the North and South Canadian Rivers of Oklahoma are relatively clear at low flow. However, a few kilometers away, e.g., in the Walnut River at Winfield, Kansas, or the Washita River in southern Oklahoma, turbidity levels are very high. The Washita River, which drains erosible red soils of southwestern Oklahoma, is the most turbid river I have encountered. Similar differences in turbidity are apparent in northern prairie rivers in Montana, Wyoming, and South Dakota: the Yellowstone, Big Horn, and Tongue Rivers are clear, whereas the Powder and Cheyenne Rivers are highly turbid with quicksand bottoms (Gore, personal communication).

Prairie mainstreams and creeks do not differ merely in transparency. Dissolved solids may differ dramatically across short distances. The Red River flows in western Oklahoma across deep Permian salt deposits from which ions are leached to the surface by numerous springs and small creeks. As a result, its salinity is higher than that of neighboring rivers, with conductivity approaching that of sea water. In those conditions only two native fish species are able to survive (Echelle et al. 1972). Near the town of Oscar, Oklahoma, a small creek flows with clear water about 4 km from its salt-spring source to the Red River. The salinity of this beautiful little creek exceeds that of sea water much of the year, and the ichthyofauna is dominated by salt-tolerant pupfish. When spates or high waters dilute the stream, fishes temporarily invade from the river, but disappear from the creek as salt concentrations return to normal (C. Hubbs, University of Texas and D. Edds, B. Wagner, Oklahoma State University, unpublished data).

Historical perspective

To have a frame of reference for considering prairie streams as natural, potentially coevolved systems, we need to know how much these streams now differ from their pristine state. We know that vast differences in the prairies followed settlement: fires no longer held back encroachment of trees in some areas (Leopold 1949); agricultural activities deeply overturned the soils and introduced many artificial chemicals into watersheds; cattle replaced the native bison as dominant grazers; and gallery forests were cut to provide firewood or lumber. In the face of such alterations, how different are prairie streams in 1988 compared with those in 1788?

Metcalf (1966), reviewing studies by historians and anthropologists, suggested that droughts sufficient to depopulate the plains occurred repeatedly in prehistoric times. As early as 1800, considerable variation existed in current or depth of streams in the Kansas River basin, as suggested by journals of Lewis and Clark (Metcalf 1966). Turbidity appears to have been quite variable; Thomas Say reported the gallery forest of the Kansas River "about a half-mile wide, but not entirely uninterrupted". Fremont found the lower Blue River in Kansas to be a "clear and handsome stream ... running with rapid current" (Metcalf 1966). Fremont also found some creeks in the region had clear water and sandy beds, but that others were dry. Metcalf (1966) noted that none of the early explorers in the Kansas River basin reported silty bottoms, although several mentioned sand or quicksand. Metcalf summarized that "the Kansas River and its larger tributaries have long been subject to fluctuation in amount of discharge. There seems also to have been considerable fluctuation in the past, as now, in the degree of turbidity, especially in the larger streams". Metcalf noted that observations by the naturalist O. P. Hay in 1885 suggested that stream siltation was notable only in eastern parts of streams of northwest Kansas, but that in the 1900s siltation progressed farther upstream, owing to erosion attributed to agriculture. Metcalf (1966) noted depletion of numerous fish species since the time of Hay, and that certain gastropods have become extinct in the Kansas River system in this century. Cross and Moss (1987) documented negative changes in fish faunas of the Kansas prairie streams, noting that small-stream fish faunas disappeared or suffered declines first, and that in the last 30 yr even fishes generally tolerant of "big-river" conditions have declined markedly.

Isaac Cooper was a member of the Fremont expedition as it traveled across Kansas to Colorado, south through the present Raton Pass of New Mexico, then eastward along the South Canadian River to Ft. Gibson, Oklahoma (Mower and Russell 1972). Of streams in the Kansas River basin (June 1845) he wrote: "we found pretty good water—there exist some fine springs along these streams and the finest water I ever drank was out of the spring called Big John. For the most part however, travellers have to drink rain water ... though this stream (Arkansas River) be a mountain torrent and flows from snow covered peaks, yet owing to its wide channel at this place and loose, sandy bed, there was barely sufficient water in it to flow. The sun's rays had full control over it and it was warmer than fresh milk during the day". One cannot help being impressed that this description fits much of the mainstream Arkansas River today, upriver of its conversion to the Kerr-McClellan navigation channel by the U.S. Army Engineers.

The Smoky Hill River (Kansas) was described by Cooper as a "broad channel filled with muddy and thick water ...", and, "its water is of a dusky and smokey colour being well impregnated with clay & sand" (Mower and Russell 1972). In contrast, in field notes of 14 June 1978 I described the Smoky Hill River as "clear to slightly turbid", over a bottom of sand and small gravel. Thus, in 1845, before plowing or grazing in the western plains, some streams like the Smoky Hill were apparently highly turbid or silt-laden during some periods.

In the upper South Canadian River valley of the present-day Texas panhandle, Cooper found "several small creeks of good water", and "a beautiful spring". The waters of the Canadian River he found "impregnated . . . strongly with red clay & sand" (Mower and Russell 1972), much as it exists today. He also wrote: "the course of the Rojo (=Canadian) is too variable; alternately winding like a serpent from one side to the other & when elevated by freshets . . . overrunning the whole basin". This description of the river channel applies very well now to the South Canadian River. At another location, Cooper reported the South Canadian River channel "upwards of a mile in width, and over the whole of this space, there was flowing but a small quantity of water in the southern side, not deeper than 2 or 3 inches". He described the "peculiarity" of the Canadian and most of the mountain streams that flow across the prairie was that the bed was well-supplied with water from the mountains, but that upon entering vast beds of sand (east Texas panhandle and west Oklahoma) it is "swallowed up" (Mower and Russell 1972).

On 3 August 1853, Lt. A. W. Whipple found the Canadian River, 40 km above its junction with Arkansas, was about 120 m broad (Foreman 1941). "The water flows sluggishly; is of whitish color, nearly clear, and less than knee-deep". Smaller streams described by Whipple on 16 August 1853, were "but a thread, winding through a gravelly bed thirty feet in width . . . showed water only in pools", much like many prairie creeks today. Slightly southwest of Purcell, Oklahoma, Whipple found numerous springs (none of which are now known to me to exist). On 25 August 1853, Whipple described tributaries of Walnut Creek south of Purcell, Oklahoma, as "Many rivulets, with crystal waters dancing in the sunlight . . . several branches, whose clear depths afforded new varieties of fishes . . .". None of Walnut Creek today is as clear as the stream Whipple described. On 27 August 1853, Whipple found "the lovely valley of Deer Creek, which bears the clear sweet waters of numerous tributaries to the Canadian". In 1978 I recorded this creek as "highly turbid, red". On 4 September 1853 Whipple found the South Canadian River in Roger Mills Co. (west edge of Oklahoma) not a "noble stream", but flowing in various small channels over a bed about 150 m wide, and "red with mud".

Cpt. R. B. Marcy visited the Little Wichita River (Clay Co., Texas) in early May 1852 (Foreman 1937), stating that "The stream at fifteen miles above its confluence with Red River is twenty feet wide and ten inches deep, with a rapid current, the water clear and sweet". In June 1981, I described the same stream as muddy, with high turbidity, and a bottom of slimy mud. On 9 May 1852 March found the Big Wichita River (Clay Co., Texas) deep, sluggish, about 50 m wide, and the water at high stage very turbid, being "heavily charged with red sedimentary matter". The Red River was 120 m wide, 2 m deep at high stage, "highly charged with a dull-red sedimentary matter, and slightly brackish to the taste". In west Oklahoma, near North Fork of Red River, Marcy's party often found clear springs of "cold, limpid water" (Foreman 1937), which are rare or absent in that region now.

The writings of the early explorers, describing streams in Kansas, Oklahoma, or Texas before any substantial settlement, provide us with information on physical characteristics of the prairie streams before the advance of agriculture and watershed modification. My overall impression is that the pre-settlement mainstream prairie rivers, like the Kansas, Canadian, or Red were in many characteristics much as they are now, with irregular or braided flow over wide beds, and periods of high turbidity. Silt apparently increased with advent of agriculture (Menzel et al. 1984, Metcalf 1966) and there is no doubt that changes in river mainstreams have negatively affected fish faunas (Cross and Moss 1987). However, we should also consider the possibility that large-scale spates in prairie rivers with low relief and unconsolidated substrates may have frequently changed meander patterns, bedload, erosion, and suspended load conditions, both in historic and pre-historic time. Even a single 100+-yr event might result in significant changes in channel geometry, with resulting differences in suspended load or water chemistry.

The greatest discrepancies between the streams described by the early explorers and those of today seem to be in the tributaries. In the journals of Marcy, Whipple, and others, many creeks described in the 1850s as clear or free-flowing are today turbid, intermittent streams. The prairie streams probably show more overall impact of post-settlement alterations than do streams of surrounding uplands. However, even in rugged parts of the Ozark uplift, streams have not been immune to changes in basic hydraulic regimes due to cultural modifications of watersheds. Black (1954) vividly described how clearcutting and conversion of Ozark hillsides to pasture can change clear perennial streams to ones flowing only half the year, and subject to extremely high discharge after rains.

We can only speculate about the degree to which the biota and ecological processes in prairie streams now resemble those of the last century, but should be alert to the possibility that "adaptations" of biota to prairie streams may have evolved in systems with characteristics or flow schedules somewhat different from those of today. If, for example, one used seasonality of flow or physicochemical conditions to explain insect or fish life cycles in an extant prairie stream, how might conclusions be altered if we knew that two centuries ago the physical characteristics or the hydrologic regime of the stream were vastly different from present?

Ecological characteristics of prairie streams

Hydrologic regimes

One important feature of streams throughout the central and southern plains is their general seasonality of flow due to common patterns in climate and geology. Although the Osage Plains to the east are wetter than the more western Great Plains, this entire prairie region is seasonal in rainfall and evapotranspiration and, therefore, in stream flow. Northern Great Plains rivers probably have a very different hydrograph, and flow more continually than do prairie rivers in Kansas or Oklahoma (Gore, personal communication). Many other parts of North America have seasonally flowing streams. The hydrograph in the west (and the northern Great Plains rivers) is strongly correlated to snowmelt and depth of the montane snowpack. In the north, winter precipitation is retained as snow, and released to swell streams with spring thaw. In the desert Southwest, stream flows are highly seasonal; some exist only as flash floods (spates) during the late summer "monsoon" (e.g., in southern Arizona). In the Interior Highlands (Robison 1986), streams fluctuate less than those of the prairie, but most Ozark or Ouachita streams show some seasonality, with high flows in spring–early summer.

Streams of the southern prairies have seasonal features that are somewhat like those of low tropics of both hemispheres. In lowland tropics of Panama (Zaret and Rand 1971) and southeast Asia approximately half the year is extremely wet and half very dry. In the southern plains of Vietnam slightly north of the Mekong delta, a dry period from approximately August to January desiccates small streams. Early in the new year rains begin, typically with heavy downpours late each afternoon. The previously dry countryside becomes a quagmire, and streams run full (Matthews, personal observations). In spite of the generality of tropical wet-dry seasons, streams of the low tropics have flow patterns that vary considerably and unpredictably among years, thus it may be difficult for stream organisms to adapt to any specific temporal pattern (B. Statzner, personal communication). Does similar low predictability apply for streams and organisms of the American plains?

Streams of the central and southern prairies of North America have a distinct wet-dry cycle, with heavy rains in spring and early summer. After mid-summer, evaporation is rapid and many prairie streams are subject to annual desiccation from late summer through winter. Much of the prairie receives rain in late summer that on average almost equals that of spring or early summer. In Tulsa, Oklahoma (96°W latitude) all months from March to October average 7.6 cm of rainfall (U.S. Department of Agriculture 1941). Weatherford, Oklahoma (99°W latitude) averages >5 cm monthly of precipitation April to October. For all of Oklahoma, mean precipitation from 1886 to 1938 averaged >7.6 cm monthly from April through September, with only November through February having <5 cm (U.S. Department of Agriculture 1941). Desiccation of prairie streams during late summer probably relates more to transpiration and evaporation due to summer heating and insolation than to actual lack of precipitation. By late summer, prairie soils are very dry, and little of the rain that does fall actually reaches a stream bed.

G. R. Marzolf (personal communication) suggested that understanding movement and distribution of water is the requisite first step in any synthesis of a prairie ecosystem. He envisioned the driving variables as (1) the mid-continent hydrologic regime, (2) solution processes in soil and groundwater, and (3) riparian vegetation. Elucidation of these complicated interrelationships is one of the goals of the Long Term Ecological Research (LTER) studies at Konza Prairie (Kansas). Marzolf noted that the location of tallgrass prairie coincides with the dividing line between positive and negative mean annual precipitation-to-evaporation values. To the east, in tallgrass prairie, there is enough water to support tree growth, but prehistoric prairie wildfire was sufficiently frequent to prevent establishment of trees.

In the prairie studied by Marzolf most streams are intermittent, and water flux depends upon lateral versus downward movement of groundwater. Konza prairie has an annual water budget of 820 mm, with approximate outputs of 180 mm lost to groundwater, 450 mm evapotranspiration, and 190 mm for streamflow (Marzolf, personal communication). In the Konza Prairie four gauging weirs and many precipitation gauges facilitate an ongoing study of water flux in unburned prairies, versus those burned at varying intervals, and will soon permit assessment of effects of native bison (Marzolf, personal communication).

Apart from local vagaries in climate and streamflow, the central and southern plains are characterized by streams that are water-limited at least part of the year. The biota must be able to survive potentially harsh conditions related to highly seasonal cycles in flow, punctuated by occasional brief but extreme spates that can occur at any time of the year. Although many prairie mainstreams now have flow partially regulated by impoundments, there is still much evidence of strong seasonality of flows. In Table 1, streams relatively unaffected by any obvious disturbances include the Smoky Hill River, Walnut Creek (Kansas), Walnut Creek (Oklahoma), Beaver River in the Oklahoma panhandle, and the Canadian River. Flow differs markedly among these selected streams owing to their east-west location and overall stream size, but all show substantial increases in discharge in May-July, and decreases from August through winter.

In contrast, note flow patterns (Table 1) at three stations on the Arkansas River in west Kansas from Lakin (westernmost) to Great Bend (easternmost). At Lakin, flow is modest in the stream channel year-round. At Dodge City, the Arkansas River is essentially dry, but at Great Bend a few kilometers downstream with inputs from small tributaries it becomes a substantial stream with year-round regulated flow. The U.S. Geological Survey notes that for the Arkansas River at Dodge City, "natural flow of stream affected by transmountain diversions, storage reservoirs, power developments, ground-water withdrawals and diversions for irrigation, and return flow from irrigated areas". Clearly, in planning any studies of prairie mainstreams, investigators must take into account a myriad of cultural factors in addition to natural climate and geomorphology.

Substrate-water interactions and deep interstitial biota

Southern and central prairie streams may differ from stony-bottomed upland streams in the extent of soil-water interactions, or the degree to which flowing water is exchanged with deep substrate. Many prairie streams can be separated into two classes: those with a primary sand substratum versus those with clay soil as the bed. Both types of substratum differ markedly from that typical of stony upland streams, although in some uplands, e.g., the Ozarks, streams may tend toward sandy bottoms. (Many northern Great Plains rivers also have cobbled substrata over most of their length, and thus may have more water exchange with the stream bed.) In clay-bottomed prairie streams, water has long residence time in individual pools as stream flow decreases in late summer. In isolated pools or in pools minimally connected by small trickles, a long residence time probably allows water and soil to come to chemical equilibrium. Such pools become in effect small ponds, with sharp thermal differences as much as 9-10°C from surface to bottom (Matthews, personal observation) and perhaps more lentic than lotic characteristics. Soil-water interaction

TABLE 1. Total monthly discharge (cubic feet per second) for prairie streams, October 1981 to September 1982. Information from U.S. Geological Survey "Water year" reports for the states.

Stream (Town, State)	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep
Smokey Hill (McAllaster, Kansas)	0	0	0	0	0	5	9	589	670	741	0.11	0
Walnut Creek (Rush Center, Kansas)	0	0	0	0	16	747	0.7	233	380	287	0	0
Arkansas River (Lakin, Kansas)	95	199	346	344	779	557	129	365	112	3899	101	2980
Arkansas River (Dodge City, Kansas)	0	0	0	0	0	0	0	0	0	11	2	0
Arkansas River (Great Bend, Kansas)	1046	1441	1303	1073	1158	3355	1267	2220	1057	3520	554	183
Walnut Creek (Purcell, Oklahoma)	931	1352	371	477	487	743	927	15,001	2901	1040	357	414
Beaver River (Beaver, Oklahoma)	0	17	3	4	6	27	26	1499	3185	987	802	0
Canadian River (Purcell, Oklahoma)	88	358	4002	736	598	3493	1143	2265	12,016	12,429	4257	1498

is also increased as many prairie streams flow over highly erodable soils, and at elevated flow carry large loads of soil particles. Within the water column, these particles can interact with dissolved ions, and also adsorb algal cells and increase their precipitation from the water column. Thus, I postulate that a clay-bottomed prairie stream is a system in which a relatively impermeable substratum minimizes water exchange to deep sediments. If the water in these streams is effectively sealed from exchange with deep sediments, there is less potential importance of water and biotic exchange with the deeper strata. The active biofilms that may be important deep in substrata of stony streams (Bretschko and Klemens 1986) may be of minimal importance or virtually nonexistent in clay-bottomed streams of the prairies, but I know of no studies addressing this question.

In prairie streams with shifting sand substrata or in stony northern prairie streams, water might be exchanged extensively through the substratum, and an active deep psammon could exist. Alternatively, if sand particles are small and well-packed, or if the substratum is characteristically a sand-mud mixture, as in the Canadian River, water might flow only minimally through the substratum for long distances. One other opportunity for water-substrate interaction in small prairie streams is the situation in which water flowing from one perched pool to another passes mostly through gravel or gravel-sand substrata of riffles, affording time for chemical reactions with those substrata. Interaction of water with gravel, particularly in riffles, may be an important part of complex interactions that, depending upon temperature, can be important in regulating stream alkalinity (Stewart 1988). Brian H. Hill (personal communication) tested Stewart's hypothesis that gravel can abiotically remove significant quantities of phosphorous from hard-water prairie streams, but permitted live microorganisms on the gravel (i.e., without drying, thus allowing microbes to live). The gravel removed up to 42% of phosphorus from water with a higher percent removal attributable to microorganisms in perennial than in intermittent streams.

Productivity and nutrients

Comparing productivity among systems is difficult, as differences in methods, time of day or year, etc. complicate interpretations. However, prairie streams for which net or gross primary productivity measurements exist appear to be relatively highly productive systems, or to have high rates of community metabolism. Bott et al. (1985) found high rates of primary production in streams of the southern Great Plains, approaching that of desert streams. Hill and Gardner (1987a) found two prairie streams of north Texas with summer gross primary productivity (GPP) ranging from 0.7 to 7.1 g O_2 m^{-2} d^{-1}, while 24-hr community respiration was 0.6–5.3 g O_2 m^{-2} d^{-1}, and commented that these rates of community metabolism approached those for highly productive desert streams. Neel (1985) found in a northern prairie stream that occasions of respiration dominance (negative balance) were rare. In a Kansas prairie stream, Gelroth and Marzolf (1978) found the GPP and 24-hr respiration ranged 0.5–1.2 and 0.5–1.8 g O_2 m^{-2} d^{-1}, respectively. My extrapolation of values from Stewart (1987) (by multiplying by 14 assumed hours of daylight) suggests net primary production on algal-colonized tiles in Brier Creek, Oklahoma of 0.7–1.93 g O_2 m^{-2} d^{-1} in midsummer, in a reach of stream where our earlier studies (Power et al. 1985) had suggested rapid increases in standing crop of attached algae. Frances Gelwick (University of Oklahoma, personal communication) found an average net primary productivity of 0.4 g O_2 m^{-2} hr^{-1} at midday in full sunlight in April, for Brier Creek cobbles covered by dense growths of *Rhizoclonium*.

I am aware of little work documenting primary productivity in upland streams of the Interior Highlands. In one Ozark stream (Baron Fork, Oklahoma), Stewart (*in* Matthews et al. 1987) reported that stream bottoms dominated by "felts" of blue-green algae were highly productive (0.6 g O_2 m^{-2} hr^{-1}), with productivity likely enhanced by the grazing of fishes. Very high productivity values were found in Blue River, Oklahoma, an upland stream of the Arbuckle Mountains, but these high values reported by Duffer and Dorris (1966) may have been very patchy. Overall, it is premature to generalize about the productivity or rates of community metabolism in prairie streams relative to that of streams at the same latitude in the Ozark or Ouachita uplands, or the Rocky Mountains. Carefully coordinated simultaneous comparisons of this basic property of stream ecosystems are needed to provide a starting point for evaluation of similarities and differences between and within such systems.

Factors limiting primary productivity in prairie stream systems are not well-understood. Stewart (1987) found that nutrient enrichment (N+P+K) markedly increased primary productivity in Brier Creek, Oklahoma, and that grazing by fishes (*Campostoma anomalum*) increased biomass-specific primary productivity of periphyton (but reduced standing crop and productivity on an areal basis). In central prairie streams (Iowa), Burkholder-Crecco and Bachmann (1979) found that nutrient concentration was not the factor limiting densities of suspended algae. In turbid prairie streams insufficient light may limit algal growth, and the shifting nature of fine-textured substrata (sands, silts) in some mainstreams probably inhibits growth of attached algae, but I know of no documentation of the latter. At low flow in the South Canadian River (central Oklahoma), I found dense mats of blue-green algae covering the sand substratum, but this typically occurred in autumn when the stream was relatively clear. I know of no measurements of productivity from such mats in prairie rivers. Interestingly, *Campostoma anomalum*, an herbivorous minnow whose diet is predominantly algae, thrives in some turbid prairie streams. For example, I have collected large numbers of this species from the Solomons River near Logan, Kansas, but I know of no one who has determined the diet of individuals from such turbid waters.

The River Continuum Concept (Naiman et al. 1987, Vannote et al. 1980) depicts many streams as allochthonous in canopied headwaters, with coarse particulate organic matter (CPOM) inputs largely as leaf fall. Farther downstream, those streams become more autochthonous, as they widen and the canopy recedes, then still farther downstream the systems become heterotrophic if turbidity or depth inhibits photosynthesis. Prairie streams likely differ from this picture in at least two ways. First, many small headwater prairie streams are sunlit due to a lack of forest, while farther downstream where flow is more commonly perennial gallery forests and closed canopy prevail (or prevailed, before settlement by western man). However, even now most prairie streams have at least some gallery forest along their lower reaches, largely because of the impracticality of farming to the edge of stream banks. Secondly, for prairie streams that depend on allochthonous inputs for energy subsidies, the material often originates from grasses rather than trees. Gurtz et al. (1982) showed that grasses composed 57% of the direct litterfall in open prairie (although total particulate organic matter (POM) input was much greater in stream reaches with gallery forest). Many prairie streams are in close association with the tall or short grass species of original prairie, or with pasture grasses. Grasslands rapidly accumulate detritus if they are protected from fires, and this detritus can affect nutrients and POM carried to the stream by overland flow during rainstorms.

Small prairie streams are probably autochthonous in many cases. In Brier Creek and other small streams of south Oklahoma I have often observed large standing crops of *Spirogyra* that assume a floating "rope" growth form, with interwoven strands sometimes accumulating to lengths of a meter or more. Such growths are usually associated with shallows, attached to cobbles or gravel of shallow pools or riffles, and are most common in late summer or autumn when stream flows are low. Agricultural activities such as fertilizer additions may stimulate this condition. However, I have also seen similar massive standing crops of attached algae in Pennington Creek, Bryan County, Oklahoma, which is a clear-water upland stream of the rugged Tishomingo Granite formation where agricultural inputs are low relative to row-crop areas. Autochthonous production coupled with input of POM from grasses could form the basis for a food web in these streams, and might minimize the importance of leaf litter relative to its role in forested upland areas of North America.

Prairie streams may differ from wooded streams in fundamental ways with respect to nutrient inputs or recycling. Nitrogen-fixation by cyanobacteria, for example, is energy-intensive and so could proceed more rapidly in well-lighted streams than in streams where light is less intense owing to canopy cover. Cultural nutrient subsidies should not be overlooked in assessing nutrient budgets for prairie streams, and such subsidies probably exceed those of more upland regions. Vast areas of former prairie are now irrigated and fertilized extensively. Additionally, a large percentage of native prairie is now pasture, and pasturelands are often fertilized to increase growth of forage. The cattle themselves are a fact of life for ecologists studying many prairie streams. Is it logical for us to seek pasture streams with cattle excluded from study sites, when the typical range of nutrient inputs in a large percentage of prairie systems now includes inputs from cattle near or within the stream?

Microbial processing and organic matter dynamics

Microbial processing of materials may also be uniquely adapted in grasslands (Marzolf, personal communication). The research group at the Konza Prairie reserve in Kansas found bacteria of streams specialized for the kinds of substrata they most often find in nature. In laboratory experiments (Marzolf, personal communication, McArthur et al. 1985), bacteria isolated from streams within gallery forest grew well on leachate from bur oak leaves, and also on leachates from big bluestem grass. In contrast, bacteria isolated from grassland streams grew well on big bluestem leachates, but poorly upon bur oak leachates. Marzolf concluded that bacteria from forested stream reaches are exposed not only to tree leachates, but also to grass leachates, because grass dominates the slopes above the gallery forest. However, bacteria from streams on grassy upland slopes would have no occasion to be adapted to grow on products from the oak trees. Furthermore, components of some bur oak leachates were toxic to some grassland bacteria. The bacteria in the grassy uplands may be adapted to quickly process monosaccharides, which may be most available for short periods of time just after stormflows (Marzolf, personal communication). This line of investigation suggests that microbial populations of prairie streams may offer opportunities for research on highly specialized adaptations.

Little is known about functional responses of streams to intermittency, which is common in Great Plains (Hill and Gardner 1987b). Theoretically, seston dynamics (e.g., amounts, transports, rates of processing of CPOM) will differ in intermittent streams, in which shredder organisms are likely less numerous than in permanent streams (Hill, personal communication). Recently, Hill and co-workers compared seston dynamics in two intermittent Texas prairie streams. For both streams, retention was slight, as there was less than one debris dam per kilometer, and total seston concentration was significantly related to discharge. Hill and Gardner (1987b) hypothesized that a lack of retention devices distinguishes prairie streams from forested upland streams which have more debris retention dams and filter-feeding macroinvertebrates, and might account for an overall higher level of seston transport in prairie than in forested streams. Hooker and Marzolf (1987) and Tate and Gurtz (1986) found shredder insects low in abundance in prairie stream leaf-pack experiments, and that elm leaves decayed faster in a perennial than in intermittent prairie streams. Brown and Ricker (1982) also found low abundance of shredders in leaf-packs in an upland Ozark stream that is near the plains. Seston in the Texas and Kansas prairie streams was dominated by ultrafine POM (Gurtz et al. 1982, Hill and Gardner 1987b). Hill (personal communication) concluded that prairie streams appear overall to be dominated by POM of smaller size classes than POM in forested streams of upland regions.

Presumably, much of the processing in intermittent streams in the presence of fewer shredders is microbial, leading to small particulate size. The dominance of fine particulate organic matter (FPOM) in prairie streams may also result from FPOM inputs by bank erosion, wind deposition, and overland runoff (Gurtz et al. 1982, Hill and Gardner 1987b). Hill and Gardner (1987b) also reported that periphyton is a dominant source of POM in some prairie streams. Hill (1988) found in small tributaries of a fourth-order, intermittent Texas stream that litter from grasses and forbs dominated the POM, whereas farther downstream tree leaves were the primary POM source. However, these allochthonous sources influenced POM dynamics rather briefly (4 mo) after which periphyton production supported the stream ecosystem. Hill et al. (1988) also examined rates of leaf decomposition in Texas prairie streams, finding slightly lower breakdown rates at intermittent than at perennial sites. Also, within the perennial stream, breakdown rates were higher at third- and fourth-order sites than at a second-order site. Hill (1988) experimented with models derived from classic stream hydraulic parameters, designed to predict organic matter dynamics for a prairie stream; based on various studies by his group in prairie streams, he concludes (personal communication) that intermittent streams have a lower ecological stability than perennial streams.

Biota of prairie streams

The biota of prairie streams is relatively well described, but the degree of coverage varies with taxonomic group and among stream types. Fishes are likely the best-explored and best-known group. Earliest detailed investigations of fishes in prairie streams were the U.S. Army "railroad surveys" conducted in the 1850s to seek routes from Missouri or Arkansas to the west coast, which led to major taxonomic advances (and some false paths) in the 1860s. For example, the expedition led by Capt. Robert Marcy crossed west Oklahoma and Texas, en route to the west coast in 1853. Surgeons attached to these expeditions collected large numbers of plants, fishes, and vertebrates which were later described and classified by scientists at the U.S. National Museum. Excellent summaries of zoogeography of prairie fishes are by Cross et al. (1986) and Conner and Suttkus (1986), and historic alterations of the fish fauna of the central prairies are documented by Cross and Moss (1987) and Pflieger and Grace (1987). Matthews (1985) offers multivariate comparisons of habitats of numerous common prairie-stream fishes, and most states in the region have a good to excellent book on fishes by a skilled ichthyologist.

Fishes generally are more diverse in the eastern prairie, with greatest species richness in streams with both upland and lowland characteristics, stony bottoms in part, and complex structural features. Richness of fish faunas is low in the western plains, where salinity (Echelle et al. 1972), lack of water (Cross 1967), or other physicochemical harshness (Matthews 1987) may exclude many fish species. Although the fishes of the prairies have been well explored and described taxonomically, there is a dearth of ecological knowledge for many prairie stream fish communities. Some common prairie fishes offer interesting enigmas. Do *Notropis girardi* really spawn only in the midst of spates, as suggested by Moore (1944)? Are some of the small prairie fishes annuals, hatching in early summer and themselves spawning in late summer or early autumn? *Cyprinella lutrensis*, for example, appears to breed successfully in late summer, and general patterns in length-frequency of populations in the Canadian River suggested that these spawning individuals were young-of-year (Matthews, personal observation). Why were some *C. lutrensis* in this stream only 14 mm long (barely post-larval) in January of 1976? Were they in fact spawned in late autumn, just before cold weather? Little is known about life-history tactics of most prairie fishes. Some of the biggest gaps in knowledge about prairie-stream fishes are in the large rivers. The fish fauna of many large, deep prairie river mainstreams needs better quantification, and the effect of impoundments upon those faunas are poorly understood.

The invertebrates of many prairie streams have been described in detail (Davis 1980, Gore 1980, Henry 1986, Morris and Madden 1978), but the degree to which a typical prairie stream fauna differs from, say, that of upland Ozark streams (Brown and Ricker 1982, Cather and Harp 1975), awaits better detail on more streams in both regions. Neel (1985) provides a highly detailed account of seasonal and annual variation in invertebrates of a small northern prairie stream. He found lower diversity of invertebrates (133 taxa) than exists in many streams at lower latitudes. Prairie streams, particularly those that are intermittent, may have limited aquatic insect faunas. Brian Hill (personal communication) found only about 1–2% the abundance of benthic invertebrates in an intermittent prairie stream relative to the number that could be expected in a perennial upland stream of the same size. The same factors that limit species richness of fish communities likely limit richness of invertebrate communities: lack of water, unpredictable flows, homogeneous substrata, and (possibly even more critical to invertebrates) a preponderance of mud or sand bottoms in many prairie streams. Conversely a relatively rich benthos may be found in riffles of prairie streams. Finally, the invertebrate fauna of a special class of prairie streams—springs or spring runs—can be as rich in species as those of more upland, mesic, or forested regions. In a two-summer survey of 50 springs through Oklahoma, J. J. Hoover and W. B. Milstead found substantial invertebrate faunas associated with springs in prairie areas (Matthews et al. 1983).

Although invertebrates in prairie streams have in general been reasonably well-documented, many areas remain unexplored. Few studies have focused on deep infauna of prairie streams, or interstitial biota (psammon) at any depth. If, as postulated above, clay substrata inhibit water-substratum exchange, in-fauna in many prairie streams may be less than in stony upland streams (Williams and Hynes 1974), but streams with unstable sand beds might be fruitful sites for investigations of a micro-infauna. Whitman (1979) investigated the psammon community of a Texas creek, and Whitman and Clark (1982) found 2–3 ppm dissolved oxygen as deep as 30 cm in the sand substratum of the stream bed throughout most of the year.

Zooplankton may be abundant in prairie river mainstreams. Repsys and Rogers (1982) found relatively high densities of microcrustaceans in the Missouri River from late autumn to spring. While drift-netting for fish eggs in the main channel of the Red River above Lake Texoma one April, I found large numbers of cladocerans in samples. However, these collections were made during high discharge, when some zooplankton found in river mainstreams may actually be produced in ponds or backwaters and washed into the mainstream. William B. Richardson and Bret C. Harvey (University of Oklahoma, personal communication) found substantial numbers of microcrustaceans in pools of small prairie streams, including several taxa of benthic microcrustaceans such as *Eucyclops* and *Bosmina* that were formerly considered to be obligate plankton. A benthic microcrustacean assemblage can be common both in prairie streams (Brier Creek) and in nearby streams in the Arbuckle or Tishomingo granite uplifts (W. Richardson and B. C. Harvey, University of Oklahoma, personal communications).

Attached algae of prairie streams are historically less well known than are fish or invertebrates. For example, in one study of periphyton of eastern Oklahoma streams, Pfiester et al. (1979) reported 344 taxa, 115 of which were new records for Oklahoma. Power et al. (1985) found that *Spirogyra* and *Rhizoclonium* dominated the periphyton of a south Oklahoma stream, with attached diatoms and bluegreen algae almost ubiquitous as an understory. Power and Stewart (1987) described the resistance to scour and recovery of the periphyton community from spates. In this stream (Brier Creek), algae largely recovered in three to four weeks after a major scouring spate. Matthews (personal observations) found that a similar period of time was required for re-establishment of blue-green algae felts on rocks in an Ozark upland stream after flood scour. Roeder (1977) found in the Skunk River, Iowa, that planktonic algae originate primarily from the substratum, with the diatom *Nitzschia* dominating both benthic and planktonic assemblages, but that there is much exchange between the two. Neel (1985) found a substantial riffle periphyton assemblage dominated by *Cladophora* and diatoms in a northern prairie stream. Even primary descriptions of the algal assemblage seem to be lacking in many prairie streams, and temporal-spatial dynamics of the flora of these streams is very poorly known.

Biotic interactions

Peckarsky (1983), Statzner (1987), and numerous others have contrasted the importance of abiotic and biotic dominance in stream community regulation. Statzner et al. (1988) suggest that biotic phenomena may dominate in pool environments where hydraulic stress is low, whereas abiotic factors may play a more important role in riffle (high hydraulic stress) environments. As important as abiotic phenomena clearly are in many prairie streams, there is ample evidence that biotic interactions like algivory, predator-prey interactions, or competition influence structure of prairie stream communities, and strong multi-level cascading effects among components at various trophic levels have been documented. Power and Matthews (1983), Power et al. (1985), and Matthews et al. (1987) showed in Brier Creek that algae-grazing minnows (*Campostoma*) could regulate location and amount of attached algae in stream pools. Further, piscivorous basses (*Micropterus*) strongly influenced habitat use by *Campostoma*, and thus directly affected dynamics of the stream algae. Stewart (1987) showed that grazing by *Campostoma* increased the rate of primary production by algae per unit biomass, although cropping of the algae by these fishes lowered net productivity per unit area. Stewart (1987) also showed that even when algal growth was enhanced by fertilizer additions, grazing by *Campostoma* played a major role in regulating algae production. Few studies, if any, of grazing effects due to stream invertebrates have been carried out in prairie streams, but I have in progress a three-year study that will include this comparison. Further, this study will examine effects of algae-grazing minnows not only on algae, but indirectly upon benthic invertebrates, and upon a variety of system-level processes or phenomena such as productivity, transport of particulate organic matter, and nitrogen fixation in prairie versus upland streams. We have already found (Matthews et al. 1987) that some of the effects of algae-grazing minnows differ between Brier Creek and upland streams of the Ozark Mountains, and suspect that differences in the dominant piscivore are involved (Harvey et al. 1988).

Harvey (1987) showed another example of cascading effects among multi-trophic levels in Brier Creek. In observational and manipulative experimental studies, he found that many larval fish in the stream are eaten by "medium-sized" fish: minnows and juvenile sunfish. However, largemouth bass in the creek prey upon the medium-sized fish. Harvey (1987) showed that in pools where large bass kept the medium-sized fish at bay (i.e., forced them to inhabit shallow stream pool margins), larval fish were provided refuges in which they survived in greatest numbers. The larger bass protected the larval fish by eliminating the threat from the fish that preyed on them. Algivory and predator–prey control are most important in Brier Creek during relatively stable environmental conditions, and become less important during or immediately after floods or other disturbances. The Working Group at Flathead Lake discussing biotic–abiotic interactions pointed out the importance of time-scale considerations where biotic interactions are concerned. How do effects of spates or droughts compare or interact with biotic interactions to determine the ultimate community structure or dynamics of community structure of prairie streams?

I know of no clear demonstrations of competition in controlled experiments in prairie streams. However, there is circumstantial evidence that interspecific competition can and may have played a substantial role in producing the biotic communities we find in prairie creeks or rivers. Historically, the Arkansas River Shiner, *Notropis girardi*, was restricted to the Arkansas River drainage, and the very closely related Red River Shiner, *Notropis bairdi*, was restricted to the drainage whose name it bears. In the mid-1970s, *N. bairdi* first appeared, apparently by accidental transfer, in the native range of *N. girardi*. During the next decade, *N. bairdi* spread rapidly through much of the range of the formerly abundant *N. girardi*, which virtually disappeared from major portions of the system as *N. bairdi* continued its invasion (Pigg 1987). The decline of *N. girardi* has been so drastic that the fish has now been listed by Oklahoma as a "species of special concern", and it is rare in some parts of Kansas. While such circumstantial evidence does not "prove" that competition is rampant, and *N. girardi* is disappearing in some areas not yet invaded by *N. bairdi* (F. B. Cross, personal communication), it suggests investigations of common resource use by these species would be useful. For all the "toughness" that we attribute to prairie stream organisms (physicochemical tolerance, ability to find food in harsh environment, etc.), perhaps many of them are really "delicate" organisms, barely existing in harmony with an environment to which they are adapted, and highly vulnerable to change.

Physicochemical limitations on the biota

The biota and biotic processes of many prairie streams are regulated at least in part by physicochemical stress. Water limitations and desiccation set absolute limits of existence for fish, but some invertebrate taxa and many algae and microbes can survive periods of drying of the stream bed. Matthews (1987) found that fish recolonized a rewatered prairie stream within the spring or early summer of one year by movement from permanent pools, and that positive correlation existed between oxygen tolerance and ability of species to colonize. Gore (1982) followed colonization of fish and benthic invertebrates in a new channel of the Tongue River, and Smith and Distler (1981) evaluated time of recovery of benthos after a chemical discharge in a sandy plains stream. Many aquatic invertebrates arrive rapidly by flight, and algae and microbes may reappear in a rewatered stream by virtue of air transport as well as hydration of resistant structures.

Prairie streams may also be stressful environments with respect to temperatures and dissolved oxygen concentrations (Matthews 1987, Matthews and Hill 1980). In the South Canadian River mainstream near Norman, Oklahoma, I found midsummer water temperatures up to 38°C, which appeared to restrict fish to cooler microhabitats. Matthews and Maness (1979) showed a direct relationship between thermal and oxygen tolerance of four minnow species and their late-summer abundance in the Canadian River. Matthews (1987) showed more broadly that numerous fish species of the genus *Notropis* from prairie streams were more tolerant of temperature and oxygen stress than were congeneric species of environmentally benign upland Ozark streams. Overall, for minnows of the genus *Notropis*, Matthews (1987) gave evidence that their success in harsh environments of prairie streams is related to physicochemical tolerance, acuity of selectivity of microhabitats, or both.

In Brier Creek, intermittent headwaters are harsh in summer relative to a perennially flowing mid- and lower section. During cessation of flow in Brier Creek, we found temperatures of 39°C in shallow pools, and documented direct heat death of fish in the stream at that time (Matthews et al. 1982). In August 1982, continuous temperature recorders showed average diel fluctuations of 9–10°C in Brier Creek headwaters, while temperatures fluctuated only 1–2°C in the midreach (Matthews 1987). At this same time, early-morning oxygen concentration in 10 of 11 isolated pools in the headwaters was 0.4–2.0 ppm, which is low enough to stress most freshwater fish. Within Brier Creek, longitudinal distribution of the common fish species was positively correlated with their tolerance to low oxygen conditions.

Another way to deal with stress, if the environment is predictable, is to synchronize life cycles to avoid exposure of vulnerable life stages to stress. In many parts of the world aquatic insects adapt, even among populations, to univoltine versus multivoltine reproductive patterns, depending upon seasonality of the environment. Bernhard Statzner suggested (personal communication) that investigation of life cycles of invertebrates in these harsh prairie streams, and comparing similar taxa among prairie, upland, etc. would be a fruitful area of investigation.

Disturbance

Environments like prairie streams that undergo stress from climatic extremes or from abrupt changes in water chemistry could be thought of as subject to frequent disturbance. However, caution must be exercised in defining "disturbance" (Resh et al. 1988—see this issue). In prairie streams of the central and southern plains, physicochemical extremes are generally predictable over annual cycles based on meteorological data. If hot, low-oxygen conditions occur every year in drying stream channels, the organisms might become adapted so that these apparently stressful events are not really a disturbance. On a shorter time scale, a predictable diel cycle of stress in many prairie mainstreams and tributaries results in lowest oxygen in early morning hours, and highest temperatures in midafternoon. Again, aquatic organisms are more likely to adapt to such predictable episodes than to events that occur erratically. Resh et al. (1988) stressed that aquatic biota of the prairies are likely adapted to allow for a certain variability about some mean measure of potential stress. Events that fall within the expected range would not constitute a disturbance, but events outside the expected range (perhaps determined by mean ± 1 SD, or some such convention) might constitute a disturbance.

Duration as well as intensity plays an important role in determining whether or not a given event is a "disturbance". Overall, disturbance would be an event that alters community organization or function, and allows for restructuring of the community. Parameters important in evaluation of a potential disturbance include intensity, frequency, duration, predictability, season, and the geomorphological setting. Measurement of the disturbance would be accomplished by evaluating recovery time for stability in levels of productivity, or stability in (or similarity to) diversity of the previous community. Although numerous authors, e.g., Ross et al. (1985) and Power and Stewart (1987) have evaluated particular disturbances in prairie streams, no single study has exhaustively examined all of the parameters above with respect to a given disturbance event in a prairie stream. Ross et al. (1985) did find that recovery of a stream fish community (Brier Creek, Oklahoma) was rapid (within a year) following extreme drought.

Droughts may be a more drastic disturbance in prairie streams than floods, if the droughts are prolonged. In the 1950s three to four years of extreme drought throughout the southern Great Plains were coincident with changes in some fish communities (Hubbs and Hettler 1958). Matthews (1987) noted the destruction of a headwater fish community in two successive years due to drought. Spates, in contrast, seem to be have somewhat short-lived effects in prairie streams. I have documented (unpublished) the composition of the Brier Creek fish assemblage by snorkeling a 1-km reach eight times, year-round. Within days following a severe spate in June 1983, the distribution of adult fishes in this stream reach was similar to that before the spate, despite the physical severity of the event (which was documented by Power and Stewart 1987). Harvey (1988) made a detailed study of washout of larval fish during high discharge in Brier Creek in early summer. While flooding virtually eliminated minnow and sunfish larvae <10 mm long, adults resumed reproductive activity rapidly after the spate. In a small upland stream not far from the prairies, Gelwick (1987) found adult fish little affected by a severe autumn spate, and that recovery of fish assemblages from high-discharge effects was rapid. The macrobenthos may be more severely disrupted by spates. Neel (1985) found macrobenthos of a prairie stream largely removed by spring spates, resulting in low population densities in late spring. The critical variables are likely event duration and mobility of organisms. Fish may find short-term hydraulic refugia during spates, but have no analogous refuge in drought. Invertebrates, less mobile than fish, may lack the ability to move to hydraulic refugia during substratum-moving spates. Clearly, adaptation of life cycles of invertebrates and fish and their life-history tactics in intermittent prairie streams need to be better understood with respect to flood and drought and the physical and chemical changes during these events.

Not all phenomena that constitute disturbance are predictable. During one late summer episode of dewatering, I marked for identification more than 700 sunfish, minnows, and bass that were crowded into two deep pools in a headwaters reach of Brier Creek. Unfortunately, rains were delayed, the pools dried up, and all the fish died. If rain had fallen a week or two earlier than it did, many of these fish would have survived and been potential colonists for that reach of stream. Perhaps the term "stochastic" applies to some aspects of ecology of prairie streams, and perhaps not. Overall, it may be difficult in prairie streams to answer a question like, "are droughts or spates worse disturbances?" However, my overall impression is that although high discharge displaces adult fish, destroys algae and invertebrate communities, and harms immature fishes, recolonization proceeds rapidly when peak discharge passes. Drought, however, kills more individuals (at least of fish) and even after dewatering, reestablishment of a biota must await colonization or regrowth from dehydrated propagules. Further, drought or dewatering seems more likely to eliminate critical components of the system, such as microbes, than does high discharge. However, after even extremely severe spates in which the entire substratum moved, I have found viable algae upon gravel (e.g., within rugosities) which can rapidly reestablish (within weeks) an active flora.

Other disturbances that need consideration in prairie stream systems are human disturbances related to impoundments, agriculture, lumbering, urbanization, mining, and so forth. The impacts of agriculture (Menzel et al. 1984) are likely to be the single most prominent feature of the western culture in prairies.

Slizeski et al. (1982) describe the wide variety of hydrological control measures that have been applied to one major prairie river mainstream (Missouri River), certainly representing major disturbance from the pristine state. Effects of impoundments and other cultural perturbations upon prairie stream invertebrates are beginning to be understood (Gore 1977, 1980, 1982, Gore and Bryant 1986, Morris and Madden 1978), but in many cases changes in a given system cannot be known precisely owing to lack of prealteration surveys. Prairie stream impoundments may have less effect upon streams than reservoirs in montane regions. Where there are steep valleys, dams are tall, reservoirs are deep, and epilimnetic releases are cold. Cold-water releases in many regions have altered trenchantly the biota of the tailwater streams (Craig and Kemper 1987, Hoffman and Kilambi 1971, Lillehammer and Saltveit 1984, Ward and Stanford 1979). This phenomenon has occurred in some prairie regions, with effects on the biota. For example, cold-water releases from Possum Kingdom Dam in north Texas have clearly altered the nature of the typically warm Brazos River, and have created riffle habitats below the dam not unlike riffles in rivers of the Ozark uplands. Interestingly, thermal tolerance and the enzyme systems of fishes below this dam have actually changed in the 40 years since construction of the dam (King et al. 1985, Zimmerman and Richmond 1981). In the northern Great Plains, numerous reservoirs with deep hypolimnial releases have had dramatic effects upon tailwater biota (Gore 1977, 1980). However, many reservoirs on southern prairie streams are not typically so deep, and thus may not markedly alter thermal characteristics of a river. Additionally, although prairie reservoirs may regulate flow for some distance downstream, many steams (the South Canadian River for example) do not show much influence of the impoundment far downstream from the dam. Southwestern prairie rivers are so influenced by infiltration of water into their sandy stream beds, by evaporation, by local agricultural withdrawals and deposits of water, that 50 km downstream from a typical southern prairie stream reservoir the mainstream probably has little impact from the impoundment. If this hypothesis is true, it contrasts with conditions in upland rivers like the White River (Arkansas) where flow is regular, and there are detectable differences in the river for many miles downstream from dams in the Ozark region, or with rivers of the more northern prairies where flow is persistent (Gore, personal communication).

Fire is a disturbance to streams that may be more frequent, or was more frequent prehistorically in prairie streams, than in many uplands. Prairie fires were once frequent enough to prevent encroachment of forests onto the prairie. Burning of prairie grass alters canopy interception of rainfall (Gilliam et al. 1987), removes accumulated plant litter, enhances warming of soil by insolation, stimulates bacteria, releases soil nitrogen, and increases primary productivity on the burned site (Marzolf, personal communication). All the results above affect the inputs of water, nutrients, leachates, and CPOM into prairie streams. Hence, fires perturb prairie stream ecosystems, but the perturbation has both positive and negative effects.

In severe winters, ice is probably a disturbance even in southern prairie stream systems. In shallow, intermittent prairie streams, complete freezing eliminates fish. Even if the freeze is incomplete, ice cover can cause substantial mortality of fish or invertebrates as oxygen is depleted under the ice (Johnson et al. 1982, Matthews, personal observations). Neel (1985) noted thicker ice cover on northern prairie streams in winters with frequent thaw and refreezing. J. A. Gore (personal communication) and students have recorded ice as thick as 0.75 m on a Wyoming prairie creek, and developed models to predict habitat availability under those conditions.

In all the disturbances in prairie streams, the perennial versus intermittent nature of flow in the system needs to be taken into account, as well as the fact that many large prairie streams (like the Arkansas River) arise in mountains. These montane-to-prairie systems might behave differently from systems that arise on the prairie. The geomorphology of the stream channel and the point along a stream at which an event occurs may also determine the degree of effect or whether an event can be called a disturbance. Finally, it may be extremely important to know whether disturbances in prairie streams act as reset mechanisms, or if they maintain prairie stream communities in a more-or-less perpetual state of disclimax.

Hydraulic approaches

Until recently, hydraulic approaches were given very little emphasis in studies of stream organisms and their functional responses (Statzner and Higler 1985, 1986). Statzner et al. (1988—see this issue) summarize many recent advances in application of hydraulics in stream ecology; thus, details need not be repeated here. How well do hydraulic-based approaches to the study of stream processes or biota relate to prairie streams of the southern Great Plains? Much of the applicability of hydraulics to ecology of stream organisms must assume that at least during much of the year, or in critical periods of their life cycles, flow is available and potentially influential. In a typical small prairie stream with only a very small percent of total area consisting of riffles, many organisms might be adapted to factors other than hydraulic phenomena. On the other hand, flow can be periodically dramatic in these streams (flash floods [spates], rapid stage rises), such that to persist, organisms may need to be as highly flow-adapted as organisms are in perennial streams.

Statzner suggested (personal communication) that the hydraulic patterns that are critical to invertebrates (or fish) in streams with substantial flow might be of less importance in many streams of the North American prairies, depending upon whether they are permanent, intermittent (but flowing more than 20% of the time), or ephemeral (with flow rare, i.e., less than 20% of the time). In streams with only occasional flow, or very low flow rates like many central or southern prairie streams, hydraulic patterns may be critical during relatively little of the year. In many prairie streams, particularly in pools, other hydraulic patterns and/or non-hydraulic phenomena may regulate standing crops of invertebrates and fish or their use of microhabitats.

I suspect that "minimum flow" may not apply in small streams of the prairie that normally cease to flow for a significant portion of the year. A critical factor may be length of time between flows, as pools shrink and animals are forced together spatially, or as quality of habitat fails: temperatures rise and oxygen declines. What about animals normally in riffles, like darters, forced into pools with predators? We have no idea how this affects them. We also have no idea how decreased flow, zero flow, drought, etc. affect phenomena such as competition between species. For example, in Brier Creek how do interactions among minnow or sunfish species change from early spring when flow is substantial to late summer when much less water is available?

What kinds of models make best predictions about hydraulic effects in prairie streams? Predictors for benthos include drift distance models (McLay 1970), benthic density-hydraulic environment models like the Gore-Judy habitat models (Gore and Judy 1981), and Statzner's hydraulic system models (Statzner 1981). For fish, at least two approaches include optimum swim speed models and stream position-net energy gain models (Fausch 1984, Fausch and White 1981, Trump and Leggett 1980). The degree to which these models apply to intermittent prairie streams remains unknown.

Areas for investigation

Some of the specific questions about prairie streams that were raised during the workshop are outlined in the preceding sections. Other questions that were suggested by workshop participants included the following:

(1) How different are stream biota and processes in perennial prairie streams, in intermittent streams that flow 20 to 80% of the time, and in ephemeral or "interrupted" streams that flow less than 20% of the time, and during much of the year are dry or exist as a series of pools?

(2) How do washout rates and retention times differ in upland and typical prairie streams, where debris dams are largely lacking? Does the long retention time of water in prairie stream pools (by virtue of low flows or cessation of flow) play a major role in nutrient transfer and all organic matter processing?

(3) How variable are nutrient inputs from rainwater (e.g., nitrate) throughout the Great Plains? Significant concentrations of nitrate seem to be in free-falling rainwater at the Konza Prairie site, but concentrations are less in parts of Oklahoma; comparative studies seem needed.

(4) Can hydraulic parameters, geomorphology, and physicochemical measurements be incorporated into a useful hierarchy of prairie stream classification that includes variables like slope, channel morphometry, stream density, etc., to facilitate broad comparisons within and among regions? Can expanded use of some of the U.S. Fish and Wildlife, Environmental Protection Agency, or U.S. Geological Survey systems be helpful? How effective is a classification based solely on mean annual discharge per unit of drainage area, perhaps with information on geology or rock type included? How should hydraulic parameters be included in stream classification schemes?

(5) How do precipitation-dominated versus rock-dominated systems (e.g., in Rocky Mountains, with nutrient limitations) differ?

(6) How do biogeographic phenomena control taxonomic and functional composition of prairie stream communities? Do biogeographic influences upon distributions need to be incorporated in schemes that seek to classify stream communities?

(7) How important are considerations of species diversity in prairie steams, not only taxonomically, but from the perspective of functional groups? What are appropriate categories of "functional groups" for prairie stream invertebrates, or fishes?

(8) Do droughts increase heterogeneity in

prairie stream systems (e.g., as pools become isolated), whereas high discharge integrates or homogenizes the whole watershed?

(9) How do bacteria make a living in prairie streams? What regulates microbial metabolism in prairie streams?

(10) What are the dynamics of the degradation side of metabolism in prairie streams? What are the overall dynamics of the dissolved organic carbon (DOC) component in prairie streams?

(11) How do root exudates contribute to nutrient inputs? How do dissolved materials from roots of streamside vegetation get into stream waters?

(12) To what extent do animals and biofilms penetrate into sediments of prairie streams, and how important are they ecologically in these systems? Do spates that scour stream beds and churn sediments result in destruction of algae and bacterial films?

The preceding view of prairie stream ecology and the Prairie Streams Workshop suggest many questions about these systems or about their role in studies of stream ecology that can be summarized in four categories: (1) basic description and comparison of biotas, processes, or rates; (2) adaptations of the biota to prairie stream conditions; (3) controlling mechanisms; and (4) comparative or experimental studies of stream ecosystems within the same latitude, including streams of prairies, lowlands, and uplands.

Especially at lower trophic levels, the biota of prairie streams is poorly known. Many basic rate measurements such as input and output of nutrients or POM, productivity, or materials processing are lacking for most prairie stream systems. Therefore, within North American prairie streams much research has yet to be done at the level of initial ecological exploration. Careful documentation of community composition and of system-level rates and processes is needed if prairie streams are to be compared rigorously among themselves or with streams of forested uplands. Little is known about adaptations of the biota to the harsh, fluctuating conditions that characterize many prairie streams. Numerous phenomena such as adaptation of insect life cycles or life-history traits of many fishes are virtually unstudied in prairie streams. I suggest, therefore, that many future studies could be devoted to basic ecological description and comparison among prairie streams, or between prairie and nearby upland streams. For example, how do ecological rates and processes, or generation times of insects differ from sand-bottomed streams of the southern Great Plains, to streams of the Ozark upland, to cold rivers of the northern prairies?

As studies in prairie streams progress beyond description or documentation, an increased focus will likely be upon mechanisms. In many cases mechanisms will not be immediately apparent from comparative studies, no matter how detailed, and experimental approaches will help clarify mechanisms that underlie community structure or system processes. There will clearly be room both for controlled laboratory research and for well-designed manipulations in the field. The strongest experiments will be those that permit clarification or quantification across several treatment levels, or that detect thresholds, interactions, or indirect effects.

Finally, although much information on prairie streams is lacking, the information that does exist supports the suggestion that the high degree of diversity of stream types within north-temperate latitudes rivals that across latitudes. Teams of investigators working at carefully selected field sites within a given temperate latitude could profitably address many of the large questions in stream ecology on a comparative measurement or experimentation basis. Granted, studies restricted to one latitude have some potential limitations. For example, regardless of the elevations of sites that are chosen, locations at one latitude will have many similarities in light regime, and studies that require simultaneous differences in day length would not be possible. However, for other studies in stream ecology investigators might find it desirable to hold effects of day length as nearly similar as possible, while varying elevation, rainfall regimes, canopy cover, stream gradient, stream discharge, and the like. Such studies could include a series of field sites at one latitude in North America: streams of the coastal plains, upland streams in the Appalachians and the Ozarks, low-gradient streams of the Mississippi delta, very high-gradient streams of the Rocky Mountains or the Sierra Nevada, and streams of the prairies and plains of the North American Midwest. An array of stream types, in which investigators could attack major questions in stream ecology, would be available and valuable.

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DETERMINATION OF TOXIC CONDITIONS IN WILSON CREEK USING FISH AND MACROINVERTEBRATE SURVEYS AND ON-SITE BIOASSAYS; WILSON'S CREEK NATIONAL BATTLEFIELD PARK, SPRINGFIELD, MO

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INTRODUCTION

Wilson Creek is a tributary of the James river in southwestern Missouri that flows southwest from within the city of Springfield through Wilson's Creek National Battlefield park. A map of the Wilson Creek drainage basin and its relationship to the park is shown in Figure 1.

For years Wilson Creek has been known for its degraded conditions. A study by the Federal Water Pollution Control Administration (FWPCA, 1969) found undesirable conditions in Wilson Creek and several incidences of "fish kills" associated with the resuspension of sludge from the Springfield Southwest Sewage Treatment Plant. Since that time, studies have included assessment of the impact of septic fields on groundwater quality (Aley and Thomson, 1984) and hydrogeologic mapping to identify areas were sinkhole flooding and serious groundwater contamination could result from land development (Aley and Thomson, 1981). Water quality problems are complicated by a marked increase in urbanization within the watershed and the interaction of surface water with a complex karst groundwater system.

The objective of this study was to assess and identify environmental impacts within the Wilson Creek drainage using an integrated approach that applied a variety of biological assessment techniques. Fish and macroinvertebrate community studies and various bioassay techniques are commonly used to determine environmental degradation (EPA, 1989a; EPA, 1989b).

METHODS

From October 1988 through July 1989, fish and macroinvertebrate communities were sampled within the battlefield from Wilson Creek and Shuyler Creek (also known as Skeggs Branch). Later, in October 1989, a survey of ambient conditions throughout the Wilson Creek drainage was completed using a variety of test species, most of which are routinely used for determining acute and chronic toxicity of effluents.

Fish and Macroinvertebrate Surveys

Wilson Creek fish communities were sampled October 1988 and July 1989 using electrofishing gear and seining techniques (Fig. 1). These data was compared to those from fish community collections from regional streams physically similar to Wilson Creek, but thought to be less disturbed (Flat Creek, Finley Creek, and Roark Creek within the White River drainage, MO). Species lists were compiled for each stream site, and the relative abundance (i.e., as a percentage

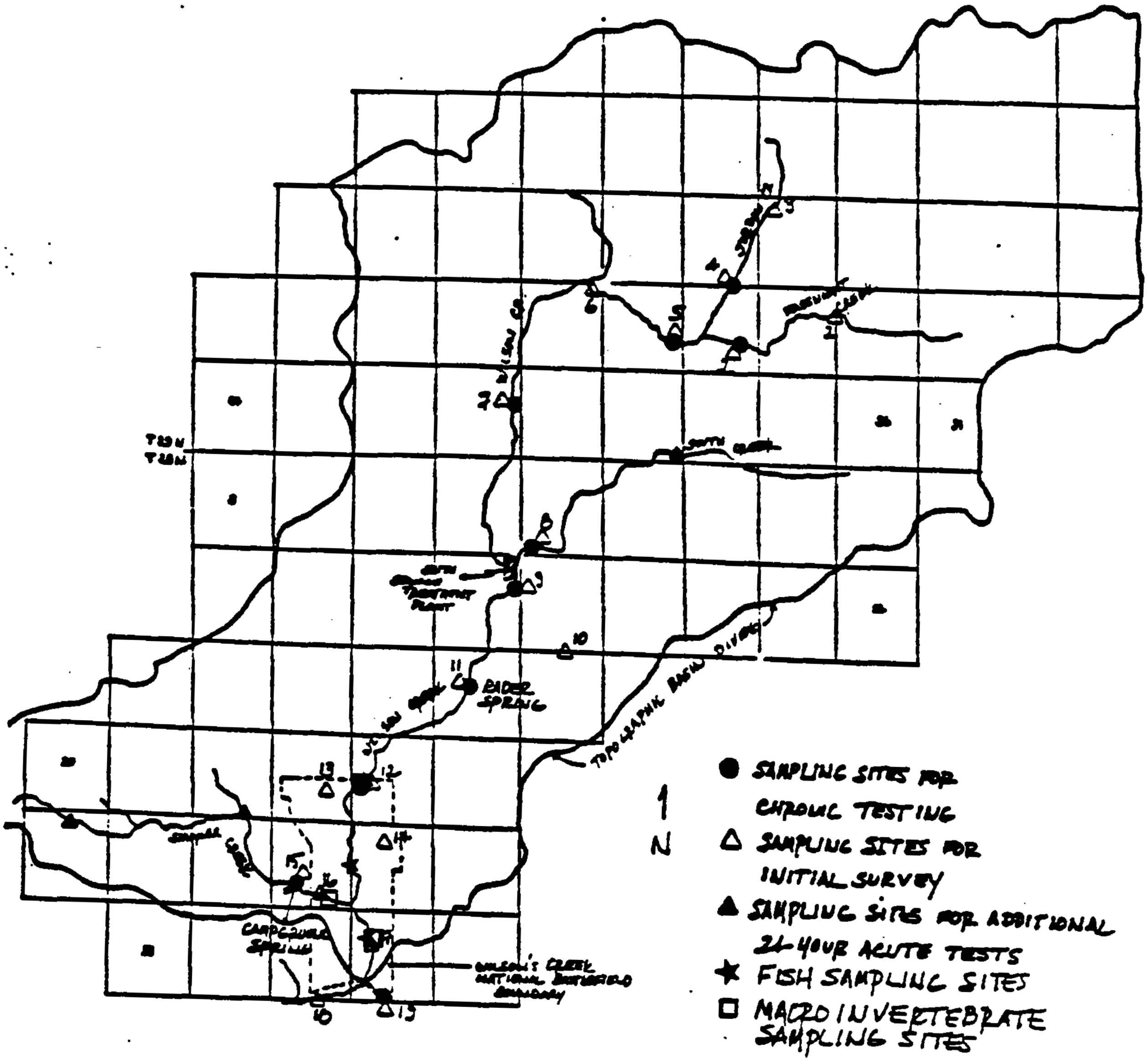

Figure 1. Map of Wilson Creek drainage basin showing location of Wilson's Creek National Battlefield park and sampling sites for all studies. Section lines are given for scale.

of total number) of pollution-tolerant and intolerant species were calculated. Pollution tolerance classifications were based on reports from regional ichthyology texts and published reports (Pflieger, 1975; Becker, 1983). Differences between the Wilson Creek and reference stream communities were further assessed using Jaccard's index of similarity (Washington, 1984).

Macroinvertebrates were collected using Surber samplers at three locations August 1988, October 1988, April 1989, and June 1989 (Fig. 1). To evaluate the health of the macroinvertebrate communities within the Wilson Creek watershed, species richness (the total number of taxa) and abundance of pollution sensitive species (EPT species, Ephemeroptera, Plecoptera and Trichoptera) at each site were determined. Community diversity was calculated using Shannon's H' (Washington, 1984), and Hilsenhoff's (1988) family-level Biotic Index (BI) provided a means to evaluate the impact of organic pollution (Hilsenhoff, 1988).

Survey of Ambient Conditions

A 24-hour acute toxicity test with Ceriodaphnia dubia was performed on water samples from 19 sites, including major tributaries, springs and visible point-source discharges (Fig. 1). Ten neonates (no more than 24-hours old) were tested in each of the 19 samples using methods of the Environmental Protection Agency (U.S. EPA, 1985).

Ten sites from the original 19 were chosen arbitrarily for further study. A seven-day chronic test of Ceriodaphnia was conducted where ten neonates for each site were transferred (static-renewal) into water samples obtained daily (U.S. EPA, 1989a). Ninety-six-hour acute tests were performed using amphipods (Hyalella azteca) and five-day old larval fathead minnows (Pimephales promelas) (U.S. EPA, 1985); as well as 48-hour barnyard grass seed (Echinocloa crusgalli) germination tests (Walsh and Weber, 1989). Dissolved oxygen, temperature, pH, ammonia, conductivity, alkalinity and hardness were also measured at these 10 sites.

Fisher's Exact Test (U.S. EPA, 1989a) was used to determine significant lethality in the chronic tests with Ceriodaphnia, while significant effects of the test water on reproduction were determined by Wilcoxon's Rank Sum Test, a non-parametric procedure used when the number of replicates are uneven (U.S. EPA, 1989a). Because no evidence of acute toxicity to any of the species tested was found, data were not subjected to analysis.

RESULTS AND DISCUSSION

Fish and Macroinvertebrate Surveys

The fish communities of Wilson Creek were dominated by a few species. Northern hognose suckers (Hypentelium nigricans) black redhorses (Moxostoma duquenei), duskystripe shiners (Notropis pilisbryi), greenside darters (Etheostoma blennioides) and stonerollers (Campostoma spp.) comprised approximately 65 percent of the total number of individuals. Reference streams were more diverse, dominated by various species of cyprinids (minnows), percids (darters) and centrarchids (sunfishes). The numbers of percids and centrarchids in Wilson Creek were noticeably less than expected.

The percentage of pollution-tolerant and intolerant species within Wilson Creek differed significantly from the reference streams. The relative percent of tolerant species found in the Wilson Creek collections ranged from 9 to 11 percent, markedly higher than the reference streams which averaged less than 2 percent. The relative percent of intolerant species, averaging 57 percent in the reference collections, was also significantly different than in Wilson Creek where only 16 to 23 percent intolerant species were found.

Similarity of fish communities differed within Wilson Creek and between Wilson Creek and the reference streams. However, Wilson Creek sites were more similar to each other (Jaccard Index 36.8) than to the reference streams (Jaccard Index ranging from 17.9 to 23.7; Table 1). The fish communities of the three reference streams were also more similar to each other than to Wilson Creek communities (median value of 31.53; Table 1).

Table 1. Community similarity scores (Jaccard Index) for comparisons between upper and lower sections of Wilson Creek and three reference streams (Flat Creek, Finley Creek, and Roark Creek).

	Wilson Cr Upper	Wilson Cr Lower	Flat Cr	Finley Cr	Roark Cr
Wilson Cr Upper	*	36.84	22.22	20.51	21.62
Wilson Cr Lower			23.68	19.51	17.95
Lower Flat Cr				35.90	27.03
Finley Cr					32.50
Roark Cr					*

The total number of macroinvertebrate species collected in Wilson Creek and Skeggs Branch ranged from 34 to 55. Plecoptera, considered to be extremely sensitive to organic enrichment and heavy metal pollution (Surdick and Gaufin, 1978), were conspicuously low in number or absent from 'collections. Additionally, the diversity of mayflies was low for a stream of this geographical area with the diversity of habitat available (Dieffenbach and Rych, 1976). The number of pollution sensitive EPT species expected in a typical Ozark Highland stream is between 18 and 22 (Dieffenbach and Ryck, 1976). Only 8 EPT species were found in Wilson Creek and 15 in Skeggs Branch.

Hilsenhoff's family-level Biotic Index calculated for macroinvertebrate collections from Wilson Creek and Skeggs Branch averaged 5.50 and 5.67, respectively (Table 2), indicating that fairly substantial pollution was likely (Hilsenhoff, 1988). The BI index suggests that both streams are receiving enrichment from upstream sources.

Table 2. Hilsenhoff's (1988) family-level Biotic Index (FBI) calculated on the mean of five samples each date.

Date	Wilson's[1]	Wilson's[2]	Skeggs
15 August 1988	5.15	4.83	5.70
14 October 1988	5.69	5.62	5.33
17 April 1989	5.97	5.96	5.78
26 June 1989	5.51	5.30	5.86

Shannon H' diversity index values for Wilson Creek averaged 1.68, and 2.47 for Skeggs Branch (Table 3). Typically, pristine streams in the geographical region of Wilson Creek have values greater than three (K. W. Stewart, University of North Texas, personal communication). In general, values from one to three are found in areas with moderate pollution and values of below one in heavily polluted water (Wilhm, 1970).

Table 3. Shannon's H' calculated on mean of five samples each date.

Date	Wilson's[1]	Wilson's[2]	Skeggs
15 August 1988	1.46	2.53	2.68
14 October 1988	1.95	2.16	2.58
17 April 1989	0.89	1.05	2.25
26 June 1989	1.66	1.76	2.38

Survey of Ambient Conditions

Twenty-four acute tests from 19 sampling sites using Ceriodaphnia did not indicate toxic conditions. Only three daphnids died; one each in water from the Springfield, MO, Wastewater Treatment Plant, Ray Spring (inside the park boundary) and Wilson Creek near the lower park boundary (Fig. 1). There was also no evidence of acute toxicity in the 96-hour tests using amphipods (Hyalella azteca); larval fathead minnows (Pimephales promelas); or on germination success of seeds (Echinochloa crusgalli).

Significant lethality and decreases in reproduction were both found in the seven-day chronic tests with Ceriodaphnia. Samples from five of the ten sites (Sites 8,9,12,15,19; Fig. 1), all within the lower half of the drainage basin, showed significantly greater lethality than Site 2 (control site in upper part of basin) (Table 4).

Samples from six of the ten sites (Sites 7,8,9,12,15,19) had significantly less reproduction when compared to Site 2, ranging from 0 to an average of 10 young per female (Table 4). Typically, under "control" conditions in the laboratory or in site water unaffected by toxic conditions, 20+ young per female are produced, which was the case at Site 2 - a spring which originates in a highly urbanized area in the city of Springfield. An average of only 13 young were produced per female at Sites 4 and 5, but they were not significantly different from Site 2 due to substantial individual variation in the number of young per female.

Daphnid sensitivity to various wastewaters, in-stream conditions, and toxicants is well documented (Mount and Norberg, 1984; E.P.A., 1986a; Burton et al., 1987; Nimmo et al., 1989; and Nimmo et al., 1990), and permit limits with biomonitoring requirements for over a thousand discharges were in place by the mid 1980's (E.P.A., 1986b). There is little question that Springfield's Southwest Treatment Plant wastewater was chronically toxic to Ceriodaphnia during this study.

Toxicity of water from Rader and Campground Springs (Fig. 1) was not anticipated since their sources are underground. Ceriodaphnia were possibly affected chronically by dissolved CO_2

Table 4. Results of chronic testing. Survival and reproduction of Ceriodaphnia tested in water from 10 stations in the Wilson Creek watershed, October 13-19, 1989.

Site Number	Description	Percentage Survival	Number of Young Produced per Female Daphnid
2	Fassnight Creek and Kansas	100	22.9
4	Jordan Creek and Grand	90	13.6
5	Middle Fassnight Creek	90	13.2
7	Wilson Creek and Hwy 60	70	10.0[1]
8	Lower South Creek	-0[1,2]	-0-[1]
9	Below Southwest Treatment Plant (wastewater)	-0[1,3]	-0-[1]
11	Rader Spring	90	8.1[1]
12	Wilson Creek at upper park boundary	60[1]	3.8[1]
15	Campground Spring	50[1]	<0.1[1]
19	Wilson Creek at lower park boundary	50[1]	3.4[1]

[1] Significantly different from Site 2, Fassnight Creek and Kansas - (≤0.05).
[2] All daphnids dead at 48 hours.
[3] All daphnids dead at 96 hours.

formed by dissolution of carbonates in the subsurface, since the pH values in these springs were lower than at the other stations (6.7 vs. circumneutral). However, there is no evidence in the literature that daphnids would have been affected by a lowered pH or dissolved CO_2; they survived and reproduced in wastewater within a pH range of 6.5 to 7.0 in other studies (E.P.A., 1986a). The fish and macroinvertebrate studies corroborate evidence from those using Ceriodaphnia to show that both Wilson Creek and Skeggs Branch ecosystems are seriously affected by conditions upstream.

In summary, wastewater from the city of Springfield continues to be chronically toxic based on both community studies and toxicity tests, and is markedly affecting aquatic life in Wilson Creek flowing through the park. Chronic toxicity found in tributaries and springs elsewhere in the watershed confirmed that both point sources of contaminants such as the Springfield wastewater treatment plant and non-point sources of unknown contaminants exist and contribute to environmental degradation.

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Fish Communities as Indicators of Environmental Degradation

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Abstract.—The basis for using biological monitoring of fishes to assess environmental degradation is that the relative health of a fish community is a sensitive indicator of direct and indirect stresses on the entire aquatic ecosystem. The most common approaches to such assessment of environmental degradation involve use of (1) indicator taxa or guilds; (2) indices of species richness, diversity, and evenness; (3) multivariate methods; and (4) the index of biotic integrity (IBI). Indicator taxa are useful when only presence–absence data are available for fish species, but they are difficult to choose objectively, their sensitivity may vary due to other factors, and they cannot indicate the relative degree of degradation. Entire trophic, habitat, or reproductive guilds appear to be more useful indicators. Determination of species richness also requires only presence–absence data, but depends on sample size, conveys little information about the community when used alone, and varies by region and stream size. Species diversity and evenness indices are not recommended because of their many theoretical, statistical, and practical flaws. Similarity indices and multivariate procedures can be used to analyze either presence–absence or quantitative data. Multivariate methods can evaluate community structure or function or both and are useful for exploring data and generating relevant hypotheses for further tests. However, they have rarely been used to detect degradation with data on fish. The disadvantages of these methods are that they are quantitatively complex, few standard procedures exist, samples used to calculate multivariate results define the multivariate space, and results can be artifacts of the procedures or data used. The IBI is a composite index that integrates attributes of communities, populations, and individual organisms to assess biological integrity on the basis of accurate measures of relative abundance. Its main advantages are that it is a broadly based ecological index, it is sensitive to different sources of degradation, and it produces biologically meaningful and reproducible results when applied by competent fish biologists. Its disadvantages are that its application requires at least moderate species richness and extensive background information and that methods for setting some criteria are subjective. It also must be modified for different ecological regions, but modifications so far have retained the original ecological framework. Future research in biological monitoring by means of fish communities should focus on (1) standardization of methods of sampling and data analysis; (2) documentation of natural variation in fish communities, against which changes due to degradation can be compared; and (3) experimental manipulation to test assumptions underpinning all the indices.

Fish communities reflect watershed conditions. This tenet—that the relative health, or biological integrity, of a fish community is a sensitive indicator of the relative health of its aquatic ecosystem and the surrounding watershed—is the basis for using biological monitoring of fishes to assess environmental degradation (Karr 1987). Karr and Dudley (1981) clarified this link when they defined

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the biological integrity of aquatic ecosystems as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region." The primary agents that stress fish communities, apart from natural environmental fluctuations, are anthropogenic disturbances. Because perturbations caused by humans often interact in complex ways, their effects on aquatic ecosystems can rarely be assessed by using only physical and chemical variables as indirect measures of biological integrity. Thus, biological integrity should be assessed directly by measurements of the aquatic biota (Black 1949; Patrick 1950; Cairns 1979; Karr and Dudley 1981; Weber 1981). In this paper we review the ways in which fish communities have been used toward this end.

Fish are useful organisms for measuring environmental degradation for several reasons. First, fish are sensitive to a wide array of direct stresses. Second, fish integrate adverse effects of complex and varied stresses on other components of the aquatic ecosystem, such as habitat and macroinvertebrates, by virtue of their dependence on those components for reproduction, survival, and growth (Karr 1981). Third, because fish are relatively long-lived, their populations show effects of reproductive failure and mortality in many age classes and hence provide a long-term record of environmental stress (Karr et al. 1986). Finally, fish communities can be used to evaluate societal costs of degradation more directly than other taxa because their economic and aesthetic values are widely recognized.

Thus, the major advantage of using fish communities to indicate degradation is that such communities integrate the direct and indirect effects of stress on the entire aquatic ecosystem and manifest the ecological significance of the perturbation. The major disadvantage is usually one of resolution. That is, if attributes of a higher level of biological organization are measured, one is generally unable to determine causal mechanisms without more detailed study.

This paper is a review and critique of the most common ways in which fish communities are evaluated as indicators of environmental degradation. We restrict our review to methods that involve fish communities and those that hold potential for such use, although investigators have taken many other approaches to assessing biological integrity, especially with macroinvertebrates (e.g., Hellawell 1977; Winget and Mangum 1979; Hilsenhoff 1982, 1987, 1988; Washington 1984). The main approaches we address are

(1) indicator taxa or guilds;
(2) species richness, diversity, and evenness;
(3) multivariate methods; and
(4) the index of biotic integrity.

We discuss the advantages and limitations of each approach and suggest appropriate methods for different circumstances, with caveats for their use. We do not address univariate statistical methods because they are treated elsewhere (e.g., Green 1979; Sokal and Rohlf 1981). For most methods, we also identify areas where further research and development is underway or is needed.

This paper is intended to aid biologists who plan to use fish communities for direct monitoring of water resource quality. Each method we discuss has strengths and weaknesses that should be considered before any method is selected as the foundation of a long-term monitoring program. We also emphasize that all methods are most useful when coupled with monitoring of physical and chemical variables, and that none were designed to replace detailed analyses of interacting biological, chemical, and physical processes nor to supplant fishery management for well-defined objectives.

Indicator Taxa or Guilds

Early investigators found that certain taxa were abundant in degraded waters, and reasoned by induction that other locations where these tolerant organisms were abundant were also degraded (Wilhm 1975). But because tolerant species also occur at undegraded sites, the disappearance of intolerant taxa has more often been used to measure degradation (Chandler 1970; Washington 1984), especially when only presence–absence data are available. Recent assessment efforts that involve fish have been based on the presence or relative abundance of individual species (Ryder and Edwards 1985) or groups of species (Berkman and Rabeni 1987; Hourigan et al. 1988). Groups of species may consist either of guilds that use the same resources (sensu Root 1967) or simply of taxonomic categories. In practice, investigators generally select indicator taxa or guilds empirically on the basis of their declining abundance or distribution with environmental degradation (Hellawell 1977; Cross and Moss 1987; Rutherford et al. 1987), rather than on the basis of experimental tests of sensitivity to specific stresses.

Examples of Use

Several investigators used fish species as indicators of organic pollution. Brinley (1942) distinguished biological zones downstream from domestic sewage outfalls in the Ohio River basin, on the basis of changes in phytoplankton, protozoa, and fish communities. He reported that "coarse" fish ("carp" and "buffalo," taxa not reported) were the first species to appear below effluents, whereas "game" fish (e.g., "bass" and "perch") replaced them farther downstream. Katz and Gaufin (1953) proposed that darters (Etheostomatinae) and black basses *Micropterus* spp. were indicators of high-quality reaches in an Ohio stream because they were the first to disappear downstream from sewage outfalls, but cautioned that their scattered zoogeographic distribution might make them inappropriate for general use. Katz and Gaufin felt it safer to rely on the number of fish species as an overall indicator. Bechtel and Copeland (1970) suggested that high numbers of bay anchovy *Anchoa mitchilli* indicated domestic and industrial sewage pollution in Galveston Bay, Texas.

Fish have also been used as indicators of agricultural degradation of streams. For instance, Hubbs (1961), Smith (1971), Greenfield et al. (1973), and Karr (1981) all proposed that the relative abundance of hybrid fishes was related to environmental degradation in midwestern U.S. streams. They reasoned that loss of habitat diversity by channelization, removal of woody debris, and siltation inhibits reproductive isolating mechanisms among stream fishes, most of which use specific substrates for spawning.

Hughes and Omernik (1981) found that two indices based on trophic guilds and tolerant versus intolerant fish species were correlated with measures of environmental degradation in midwestern agricultural streams. Menzel et al. (1984) suggested that absence or scarcity of eight "sensitive" species that feed primarily by sight on invertebrates and require coarse spawning substrates indicated stream degradation from channelization and siltation in several agricultural watersheds in Iowa. Similarly, Berkman and Rabeni (1987) found that relative abundances of benthic insectivore and herbivore feeding guilds and simple-lithophilous reproductive guilds, which require clean riffle substrates for invertebrate food production or spawning, were negatively correlated with percent silt in riffles in Missouri agricultural streams.

Other investigators used fish as indicators of more general degradation of aquatic ecosystems. For instance, Hartmann (1983) and Leopold et al. (1986) suggested that certain salmonid, percid, and cyprinid taxa indicated progressive cultural eutrophication of deep northern European lakes. Ryder and Edwards (1985) and Marshall et al. (1987) proposed lake trout *Salvelinus namaycush* as the best single indicator of ecosystem quality of the Laurentian Great Lakes, because of its position as a terminal predator in the aquatic food web and its strict environmental requirements that are met only in oligotrophic systems.

Advantages and Disadvantages

The advantages of using indicator taxa or guilds are that the approach is conceptually simple, requires no complicated theory, and can be easily applied with semiquantitative (relative abundance) or qualitative (presence–absence) sampling of fish communities. The value of indicator taxa increases when entire guilds are used because guilds allow finer resolution of stresses specific to one habitat, trophic group, or reproductive strategy.

The approach has several disadvantages, however, one of which is that there are few guidelines for choosing appropriate indicator taxa or guilds (see Landres et al. 1988 for a critique). Thus, it is often difficult to determine whether a species is sensitive to a specific source of degradation solely on the basis of empirical evidence from the field, where many factors interact to mask the effects of perturbations. Moreover, taxa may be absent or scarce for reasons other than degradation, such as zoogeographic barriers (Gilbert 1980; Hocutt and Wiley 1986), overharvesting, or biological interactions (Page and Schemske 1978; Fausch and White 1981, 1986; Baltz et al. 1982; Power et al. 1985).

Although aquatic ecologists have not yet used indicator guilds extensively, there is currently much argument among avian ecologists over their use (Severinghaus 1981; Verner 1984; Morrison 1986; Szaro 1986; Block et al. 1987). Arguments stem from Severinghaus's (1981) proposal that the effects of perturbations on single guild members can be extrapolated to entire guilds. Other investigators, however, found that guild members respond to environmental changes differently across space (Block et al. 1987) or through time (Szaro 1986) and so concluded that this assumption is dangerous. Berkman and Rabeni (1987) avoided this problem by analyzing changes in abundance of entire feeding and reproductive guilds to assess effects of siltation on stream fish communities, an approach advocated for birds by Verner (1984) and for coral reef fishes by Hourigan et al. (1988).

Another disadvantage of indicator taxa is that their sensitivity to degradation may vary regionally, seasonally, with age of fish, or for other reasons. Moreover, because sensitivity is determined by comparison to other species, it may vary with the richness or composition of the community. For example, the common shiner *Notropis cornutus* is one of the extant species most intolerant of overall degradation in the South Platte River basin, Colorado, which is at the western edge of its range (Fausch, unpublished data), but it is not considered an intolerant member of more species-rich midwestern U.S. fish communities near the center of its distribution (Karr et al. 1986; OEPA 1988).

A related point is that species may be sensitive to certain types of degradation but not others (Washington 1984; Cairns 1986). Creek chub *Semotilus atromaculatus* appear relatively intolerant of habitat degradation in eastern Colorado plains streams (Fausch, personal observation), probably because they require clean gravel for spawning, but were reported to be very tolerant of a variety of pollutants elsewhere (Katz and Gaufin 1953; Leonard and Orth 1986). Even more striking is that common carp *Cyprinus carpio* and goldfish *Carassius auratus*, which tolerate a wide variety of perturbations including high temperature, low oxygen, and degraded habitat, are among the more sensitive species tested to the organochlorine pesticides endrin and dieldrin (Mayer and Ellersieck 1986). Indeed, Cairns (1986) argued that the probability of selecting a single species that is most sensitive to all perturbations in a given situation is remote.

A final criticism is that single indicator taxa cannot be used to determine the relative degree of biological integrity because a site lacking a certain indicator species could be either slightly or greatly degraded. Thus, it is clear that indicator taxa, if used alone, are effective only for general, qualitative evaluations. Moreover, most of the information contained in fish-community data is not used in this approach. Although a single species can be used to measure perturbations at the level of the individual organism and population, it cannot be a robust measure of community- or ecosystem-level processes (Schaeffer et al. 1988). For this reason we propose that lake trout are, in and of themselves, an inadequate indicator of ecosystem quality for the Great Lakes (see also Leonard 1986). Rather than attempting to delineate appropriate indicator species for each major habitat (e.g., oligotrophic and mesotrophic pelagic zones; coastal wetlands), as Ryder and Edwards (1985) did, we suggest developing an index based on changes expected in fish communities when degradation occurs. For example, Ryder and Edwards (1985) described the shift from larger long-lived deepwater benthic invertebrate feeders and piscivores to smaller short-lived pelagic planktivores as indicative of Great Lakes ecosystem degradation during the last 200 years.

Recent Developments

In recent years indicator taxa have rarely been used alone but rather as part of more detailed evaluations of environmental degradation. Matthews et al. (1983) successfully used indicator species as a preliminary measure of habitat quality in a regional survey. Investigators now propose the use of entire guilds as indicators (Verner 1984; Berkman and Rabeni 1987; Hourigan et al. 1988); such methods should greatly reduce problems associated with single indicator taxa.

Species Richness, Diversity, and Evenness

The concept of species richness, the number of species in a sample from a community, is relatively old, although it continues to be the most basic attribute of ecological communities measured (e.g., Karr 1981; Scott et al. 1987). The concepts of species diversity and evenness (defined later) were modified for analysis of biological systems during 1950–1970. All three concepts (species richness, diversity, and evenness) have been widely used to analyze community structure (Peet 1974) and to assess environmental degradation in aquatic ecosystems (reviewed by Washington 1984). The assumption underpinning these indices is that stressed fish communities show changes in community structure—that is, changes in the number of species (richness) and their relative abundances (evenness). Environmental degradation is assumed to change "diverse" communities consisting of many species that are relatively equally abundant, to "simple" assemblages dominated by a few species.

Many indices are available to measure species richness, diversity, and evenness of biotic communities (Washington 1984). The simplest measure of species richness is s, the number of species in the sample, but it depends on sample size (Sanders 1968; Angermeier and Karr 1986).

Other investigators have attempted to ameliorate this problem by dividing s by multiples of n, the number of individuals in the sample (e.g., "species per thousand individuals"; Odum 1967; Bechtel and Copeland 1970), or by log n (Odum et al. 1960), but these measures also depend on sample size (Washington 1984).

A possible solution to this problem is to estimate asymptotic species richness by comparing rarefaction curves (Sanders 1968; Hurlbert 1971), which describe the cumulative increase in species as a function of the number of individuals drawn from the sample. In the case of stream fishes, a more practical solution is to thoroughly sample each riffle, run, pool, and backwater until no new species are captured (Hocutt et al. 1974; Hocutt 1978). The species captured after sampling several units of each habitat type should be a good measure of the fish community in a given stream reach. Moreover, if several gear types are used carefully, this sampling protocol should also yield relatively unbiased estimates of relative abundance of fish species. These estimates are required for all the methods described hereafter.

The most widely used measures of diversity, the Shannon–Wiener and Brillouin indices (Appendix Table A.1), were derived from information theory and were originally designed to determine the uncertainty in predicting the next character in a message. In biological systems, they measure the uncertainty in predicting the species of the next individual selected. This uncertainty should be low in simple communities dominated by a few species, and high in diverse ones.

The Brillouin index is used when the entire community has been collected, whereas the Shannon–Wiener index is intended for estimating species diversity from a sample drawn from the community (Pielou 1975; Kovalak 1981). The latter case prevails in most field applications. To be useful, a diversity index should be relatively unaffected by sample size after a sufficient area of habitat has been sampled to reveal asymptotic diversity. Kovalak (1981) pointed out that the Shannon–Wiener index meets this requirement but that the Brillouin index, among others, is strongly influenced by sample size and thus is useless in certain situations.

Indices of the evenness with which individuals are distributed among the species are generally calculated by dividing observed diversity by the maximum possible diversity for a community with the same number of species (Appendix Table A.1; Pielou 1975; Washington 1984). Maximum Shannon–Wiener diversity, obtained when all species have equal numbers of individuals, is calculated as log s. Various investigators have used logarithms of different bases (e.g., base 2, e, and 10) to calculate diversity and evenness, but all are related by constants so formulae can be easily converted (Pielou 1969). However, authors using these indices must define their log base and use it consistently.

Several authors modified diversity indices specifically for fish community analysis. Gammon (Gammon 1976, 1980, 1983; Gammon and Reidy 1981; Gammon et al. 1981; Gammon and Riggs 1983) developed the composite index of well-being from measures of fish abundance, biomass, and Shannon–Wiener diversities based on numbers and biomass (Appendix Table A.1). Cornell et al. (1976) calculated Shannon–Wiener diversity and evenness using as data the changes in relative abundances of species before and after environmental perturbation rather than the relative abundances of species on one date. They believed that these indices of diversity of change are more sensitive measures of environmental degradation than traditional diversity indices.

Examples of Use

Many researchers have used indices of species richness, diversity, and evenness to analyze fish community structure (e.g., Gorman and Karr 1978), but here we describe only examples in which these measures were used to evaluate environmental quality. For example, Patrick (1950) found that numbers of species of fish and other taxa were depressed at polluted sites in eastern Pennsylvania streams relative to less degraded sites, and classified site quality based on changes in species richness.

Several investigators developed relationships between diversity or species richness and chemical water quality. Bechtel and Copeland (1970) found that Shannon–Wiener diversity of both numbers and weights of fishes were negatively correlated with the percentage of toxic industrial effluent at sites in Galveston Bay, Texas. Tsai (1973) calculated regressions that described decreases in species diversity as functions of total chlorine and incremental increases in turbidity at sites downstream from sewage effluents in 135 eastern U.S. streams. Cornell et al. (1976) calculated Shannon–Wiener indices of the diversity and evenness of change in fish abundance in a Kentucky stream degraded by silt from mining pollution. They proposed that these indices, coupled with the mean absolute value of change in abundance per species, can be used as "vital signs" for a community, but most of their inferences were based on detailed analyses of changes in abundance and were supported by heuristic ecological analyses of competing species.

Gammon (1976) combined catch rates (number and biomass of fish per kilometer electrofished) and Shannon–Wiener diversity based on numbers and biomass into a composite index of well-being. He first weighted natural logarithms of the catch rates so that all four measures had similar ranges (about 0–2.75 for all) and then summed them. Gammon proposed that this combination of community structure indices is useful in detecting subtle shifts in stressed communities, has the advantage of being less variable than any of its components, and reflects major perturbations to river water quality such as sewage and heated effluents. Gammon (1976, 1980, 1983), Gammon and Reidy (1981), Gammon et al. (1981), and Gammon and Riggs (1983) used the index of well-being to assess environmental degradation in the middle Wabash River and its tributaries in Illinois and Indiana. The Ohio Environmental Protection Agency (Ohio EPA) also modified this index for Ohio streams (OEPA 1988). However, this index has been less successful than the index of biotic integrity at detecting degradation in midwestern streams (Hughes and Omernik 1981) and in the Willamette River, Oregon (Hughes and Gammon 1987).

Advantages and Disadvantages

There are several advantages to using indices of species richness, diversity, or evenness to assess changes in fish communities due to degradation. First, all have been widely used so the methods are generally understood, and there has been much previous work on theoretical aspects and statistical properties (see reviews by Peet 1974 and Washington 1984). Second, all are relatively easy to calculate, if one can assume that a representative sample of the community has been collected. Third, little background ecological information is needed to apply these indices of community structure.

Disadvantages of using species richness are that it depends on sample size, conveys limited information about the community, and varies regionally so that one must compare it to relatively undisturbed local reference sites to infer degradation. In spite of these limitations, the list of fish species collected from a site by carefully sampling all habitats can provide much information about environmental quality when interpreted by a competent ichthyologist who knows what species to expect in given habitats of a region.

In comparison to species richness, indices of diversity and evenness have many more limitations. First, species richness and evenness are mathematically related to diversity (Peet 1974), and thus diversity indices are difficult to interpret. This confounding of community attributes led Hurlbert (1971) to label species diversity as a "non-concept." For instance, a nonselective perturbation that changes a community of many species, some of which are in relatively low abundance, to one of few species that are about equally abundant may actually increase the Shannon–Wiener diversity index, even when the total number of individuals is reduced (Kovalak 1981; Karr et al. 1986). In the absence of other information, such increases in diversity might be construed as improved conditions. In light of this problem, the Shannon–Wiener index is best used to detect selective perturbations that eliminate certain species, whereas the Brillouin index can detect only nonselective perturbations such as heavy metals or other toxicants that increase the evenness of surviving species (Kovalak 1981). Most diversity and evenness indices also have other undesirable statistical properties (Peet 1975; Washington 1984).

In addition to these theoretical limitations, diversity and evenness indices incorporate relatively little biological information, which severely hampers their use in detailed analyses of ecological systems. Thus, a second disadvantage is that although these indices address community structure, they ignore the function of species in communities. Third, diversity and evenness indices do not consider species identity or absolute abundance. Although changes in numbers of species or their relative abundances influence diversity, these indices are often insensitive to species replacements or declines in absolute abundances. Indices of the diversity and evenness of change (Cornell et al. 1976) were designed to overcome this problem, but they have been little used to date, have unknown statistical properties, and require further detailed analyses to be useful.

A fourth disadvantage is that, even without degradation, fish species diversity may vary substantially by season (Dahlberg and Odum 1970; McErlean et al. 1973; Harima and Mundy 1974; White et al. 1977; Murphy 1978), among years (Angermeier and Schlosser 1987), and longitudinally in streams (Goldstein 1981). Goldstein (1981) proposed regressing species richness and diversity as functions of the natural logarithm of watershed area to account for natural increases in species richness with stream size; this analysis is similar to the analysis of maximum species richness by Fausch et al. (1984) for the index of biological integrity described later.

Fifth, diversity indices are sensitive to taxonomic resolution, although the components of diversity at various levels in the taxonomic hierarchy are additive (Kaesler and Herricks 1979). Kaesler and Herricks (1979) analyzed diversity at different taxonomic levels for selected invertebrate and fish communities and proposed that the component contributed by species was small compared to that contributed by higher taxa, but this was true for only about half of the 25 fish collections they analyzed. They suggested that time and money could be saved in environmental monitoring by identifying taxa only to genera, but we do not advocate that approach for fish.

A sixth disadvantage is that, although easily calculated, interpretation of diversity indices is difficult. After calculating the number, one must then determine how it can be used to assess environmental degradation (Karr et al. 1986). Wilhm and Dorris (1968) and Wilhm (1970) proposed rough guidelines for Shannon-Wiener diversity on the basis of empirical evidence from macroinvertebrate communities above and below sewage treatment plants; when $\log_2$ is used, a value less than 1 indicates heavy pollution, values of 1–3 indicate moderate pollution, and values greater than 3 indicate clean water. However, empirical evidence would probably suggest that these ranges are inappropriate for fish communities, which typically have much lower species richness. Moreover, diversity, like species richness, is likely to vary with stream size (Goldstein 1981) and across zoogeographic regions, so expected values are not so easily defined.

A seventh disadvantage is that, although they generally decline with severe degradation, species richness and diversity may actually increase with minor or moderate degradation. In streams tributary to San Francisco Bay, where the native fish fauna is depauperate, Leidy and Fiedler (1985) found that both species richness and diversity increased at sites subject to moderate human disturbance. Most of the increase was due to additions of exotic species, whereas the number of native species increased only slightly. This general phenomenon may also hamper other indices that make inferences from changes in species richness, especially if introduced species are included.

A final disadvantage is that although community structure is influenced by both numbers and biomass of species it is unclear which data are best to use. Although species diversity based on biomass is thought to represent the energy distribution in an ecosystem (Wilhm 1968), diversity based on numbers reflects the number of pathways open to energy transfer (MacArthur 1955). Gammon's (1976) index of well-being is an effort to combine these two measures of diversity, but we question the ecological relevance of this composite index.

Recent Developments

Indices of diversity and evenness are not as widely used or accepted as they were 15 years ago, due to the aforementioned philosophical, statistical, and practical problems (Peet 1974; Washington 1984). Methods have been proposed to ameliorate certain problems such as species replacement and longitudinal changes in species richness. Overall, however, when diversity indices have been compared to other approaches for measuring biological integrity of fish communities, the latter have performed as well or better.

Multivariate Methods

Similarity indices and the multivariate statistical procedures of cluster analyses, ordination techniques, and discriminant analyses all measure the mathematical relationships among samples for which two or more variables have been measured. For example, a common problem involves comparison of relative abundance of many species (variables) among several sites (samples). Either qualitative presence–absence data or quantitative measures of abundance or biomass can be used with most multivariate methods. Few have been used with fish community data to assess environmental degradation, but several may hold promise for some circumstances (Smith et al. 1988).

Similarity and dissimilarity indices, the simplest multivariate procedures, compare the amount of overlap in species composition or relative abundances of species between two samples. The degree of similarity or dissimilarity between the sample of interest and one or more reference samples can be used to indicate biological integrity. These analyses are most revealing when samples are compared to both relatively pristine and highly degraded reference sites. The Jaccard similarity coefficient is widely used for presence–

absence data, whereas the percentage similarity coefficient is often used for quantitative data (Appendix Table A.1). Many other similarity and dissimilarity indices exist (Boesch 1977; Wolda 1981; Polovino et al. 1983; Boyle et al. 1984; Washington 1984) but few have been used to assess fish community health.

Cluster analysis can be used with similarity measures (Cairns and Kaesler 1971) to group samples of increasing similarity (Pielou 1984; Digby and Kempton 1987). This procedure allows simultaneous comparison of the similarity of each sample with every other one; results are usually depicted as dendrograms of multivariate distances among samples.

Ordination techniques encompass a broad range of statistical procedures. They are used to order many samples, each having several variables, along a few dimensions or axes (Digby and Kempton 1987) so that similar samples are close together (Gauch 1982). The axes, which are mathematically derived from the original variables, account for the greatest amount of variation in the data and reduce the number of dimensions that must be considered. The scores of samples along these axes are presumed to be correlated with environmental gradients (such as severity of pollution or channelization) and thus are potential indices of degradation. However, interpretation of the axes may be difficult if no correlates are discovered, in which case the multivariate analysis yields little information. Ordination techniques have not been widely used to assess biological integrity with fish communities, but three techniques—principal coordinates analysis, principal components analysis, and detrended correspondence analysis (Gauch 1982; Pielou 1984)—hold some promise (e.g., Bloom 1980; Menzel et al. 1984; Berkman et al. 1986; Berkman and Rabeni 1987).

Discriminant analyses, another type of ordination, generates functions of variables that best distinguish among previously defined groups of samples (Digby and Kempton 1987). If samples from degraded and pristine ecosystems were separated into two groups and used to develop a discriminant function, scores for additional samples could be used as a relative index of environmental degradation.

Examples of Use

Jaccard similarity coefficients (JSC) have been used as indices of degradation with fish communities since the early 1970s (e.g., Richards 1976). For example, Hocutt et al. (1974) found low similarity between fish communities upstream and downstream from a Virginia synthetic fibers plant, and concluded that JSC values reflected the decline in river water quality better than species diversity indices. Cairns and Kaesler (1971) conducted a similar analysis of fish data collected near a Potomac River power plant (Maryland and Virginia) and used cluster analysis to compare similarities among stations, years, and river flows. They found that the JSCs did not differ consistently between upstream and downstream stations when differences among years and flows were accounted for, and concluded that the plant affected neither fish nor other biota. Rutherford et al. (1987) used a presence–absence similarity index in unique ways to detect changes in southeastern Oklahoma stream fish communities resulting from timber harvest.

Percentage similarity coefficients have also been applied in several studies. Haedrich (1975) found that fish communities in undegraded Massachusetts estuaries had high species diversity but low similarity among seasons, whereas communities in degraded estuaries showed the opposite characteristics. He proposed that this pattern was a general one caused by major seasonal fluctuations in species composition and abundance in relatively undegraded estuaries (e.g., Dahlberg and Odum 1970; McErlean et al. 1973) and by dominance of a few tolerant persistent species in degraded ones (e.g., Bechtel and Copeland 1970). Research in estuaries in California (Horn 1980) and Costa Rica (Phillips 1983) supported Haedrich's hypothesis. Reash and Berra (1987) also found that degraded segments of Ohio headwater streams had low fish species diversity but relatively high similarity to each other because a few tolerant species dominated, whereas undegraded reaches showed the opposite pattern due to asynchronous seasonal fluctuations in species abundances.

Ordination of fish communities has been used less than similarity indices to detect environmental degradation. Menzel et al. (1984) used principal components analysis to show that higher-quality fish communities were associated with better water and habitat quality in 10 Iowa agricultural streams. Berkman et al. (1986) and Berkman and Rabeni (1987) found that detrended correspondence analysis portrayed differences in fish communities associated with varying riffle siltation, but that it was not as useful as the index of biotic integrity for detecting changes in water and habitat quality in three Missouri agricultural streams.

Discriminant analyses of fish community data have not yet been used to assess environmental degradation, although they have proven valuable in structural comparisons of relatively undisturbed fish communities at different locations (Tonn and Magnuson 1982; Tonn et al. 1983; Rahel 1984). Larsen et al. (1986) used canonical (linear) discriminant analysis of fish communities from "least-impacted" reference streams to distinguish among four Ohio ecological regions.

Advantages and Disadvantages

All the multivariate procedures discussed have the advantages of quantitative sophistication, flexibility in data requirements, and simultaneous comparison of all samples. However, a disadvantage is that most require a computer for calculation and extensive statistical, as well as biological, training and experience to interpret.

A second advantage is that multivariate procedures can accommodate either presence–absence or quantitative data (or both) and can be used to evaluate either community structure or function. For instance, when variables are attributes of species, these procedures can evaluate richness, evenness, and species composition components of community structure (e.g., Berkman and Rabeni, 1987). Alternatively, when variables are attributes of trophic, reproductive, or other functional guilds, multivariate methods analyze community function (e.g., Larsen et al. 1986). Menzel et al. (1984) used both types of variables to simultaneously analyze community structure and function.

These examples indicate that multivariate methods are often useful for exploring the structure of data sets and generating relevant ecological hypotheses (Smith et al. 1988), a third advantage. These methods reduce the dimensions of data sets having many variables to results that can be plotted on two or three derived axes, which may clarify relationships among samples if a few of the original variables contribute strongly to these axes.

One disadvantage of using multivariate methods, beyond the requirement of statistical sophistication, is that few standard procedures exist for the various methods. Smith et al. (1988) provided a readable account of the many ways univariate and multivariate methods can be used to analyze relationships among marine benthos and environmental factors; fish biologists should find this account useful. A disadvantage of any indices derived from multivariate procedures is that they depend on characteristics of the samples used to calculate them. Because the samples define the multivariate space and the endpoints of the index, different undegraded fish communities might fall far apart on multivariate axes simply due to natural factors that cause different species richness and composition—longitudinal succession in lotic systems, for example (Sheldon 1968; Fausch et al. 1984). Therefore, in addition to data from sites with comparable fish communities, samples from both pristine and degraded reference sites must be included if such indices are to be useful.

A final disadvantage is that conclusions drawn from multivariate methods can be artifacts of the procedures. For example, researchers have been unable to find the underlying multivariate structure in simple deterministic data sets (Armstrong 1967) and, conversely, have manufactured significant multivariate relationships from meaningless data (Karr and Martin 1981; Rexstad et al. 1988). Therefore, investigators should use caution when interpreting results generated from multivariate analyses (Smith et al. 1988) and consider them as hypotheses to be tested further, preferably by experiment.

Recent Developments

Development of an index of environmental degradation on the basis of multivariate procedures requires choosing appropriate pristine and degraded reference sites. The objective criteria developed by Hughes (1984) and Hughes et al. (1986) for choosing "least-impacted" reference sites within ecological regions may be useful for this end. Alternatively, one might develop "theoretical" samples of fish communities from pristine and degraded environments.

Multivariate procedures generally were not designed for developing indices of degradation, so standard methods have not yet been developed (but see Rutherford et al. 1987, Bloom 1980, and Boyle et al. 1984 for other taxa). For example, because the multivariate relationships described previously were each calculated with different variables and fish community data, the ordination axes or discriminant functions are all different and cannot be compared. Multivariate indices would be more valuable if samples for a given region were compared to the same reference sites with the same multivariate functions (e.g., Tonn et al. 1983).

The Index of Biotic Integrity

The index of biotic integrity (IBI) was developed by Karr (1981) to assess environmental degradation in midwestern U.S. streams. It is now being applied in other regions (Karr et al. 1986; Miller et al. 1988; Steedman 1988) and modified for use in other aquatic environments and with taxa other than fish. The IBI is a composite index based on an array of ecological attributes of fish communities: species richness, indicator taxa (both intolerant and tolerant), trophic guilds, fish abundance, and the incidence of hybridization, disease, and anomalies. Thus, the IBI integrates characteristics of the community, population, and individual organism to assess biological integrity (Karr et al. 1986; Karr 1987).

The IBI consists of twelve community attributes or metrics (Appendix Table A.2) that are compared to values expected for a relatively unperturbed stream of similar size in the same ecological region. The metrics are each rated 5, 3, or 1 depending on whether their value is comparable to, deviates somewhat from, or deviates strongly from the expected value. The expected values for metrics must be set a priori by a competent fish ecologist on the basis of guidelines presented in Karr et al. (1986) and pertinent literature of, and knowledge about, regional fish communities. Criteria for species richness and composition metrics are set by means of maximum species richness lines that describe changes in the number of fish species as a function of stream size (Figure 1; Fausch et al. 1984). These relationships vary for different ecological and zoogeographic regions.

The scores for the twelve metrics are summed to yield an index that ranges from 12 to 60. From these scores, as well as other pertinent information about the fish community (Karr 1981; Karr et al. 1986), the fish biologist assigns the site to one of five classes, which have biological meaning (Table 1). A sixth class is assigned when repeated sampling yields no fish.

The IBI is a flexible index in that other pertinent ecological metrics can be substituted in regions where the structure and function of fish communities differ. A variety of metrics have been modified for different ecological regions and specific applications (Miller et al. 1988) but retain the basic ecological foundation proposed by Karr (1981).

Although the various IBI metrics are not independent, this redundancy can be viewed as a strength for several reasons. First, metrics are thought to have different, but overlapping, ranges of sensitivity (Angermeier and Karr 1986), and thus the robustness of the index is increased. For example, Leonard and Orth (1986) found that species richness and composition of darters was a sensitive metric only in better-quality streams because darters were absent from degraded sites. Site quality was measured with an index of cultural pollution that they devised. Conversely, they found increases in the proportion of individuals that were creek chub (a metric modified to indicate dominance of a tolerant species) and in the incidence of disease or anomalies only after substantial degradation was evident, indicating that these metrics are sensitive only at low-quality sites.

A second reason for use of multiple community attributes is that the relative contribution of different metrics to the total IBI score, and hence their importance in determining biological integrity for a site, varies with respect to different ecological regions, different spatial scales, and different ranges of degradation (e.g., Angermeier and Karr 1986; Steedman 1988). Angermeier and Karr (1986) suggested that all three factors were reasons why metrics accounting for the most

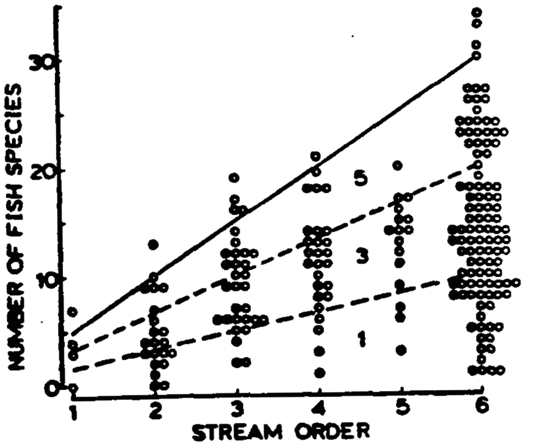

FIGURE 1.—Maximum species richness line for the Wisconsin portion of the St. Croix River watershed (after Fausch 1987). Points represent the number of species in each of 206 fish collections, all of which were obtained farther than 4.8 km from lakes or larger rivers. The region below the solid maximum species richness line is trisected by dashed lines for scoring the index of biotic integrity metric for total number of fish species. See Fausch et al. (1984) for methods of determining maximum species richness lines.

TABLE 1.—Classes of biological integrity, their attributes, and corresponding total index of biotic integrity (IBI) scores based on the sum of 12 metric ratings (modified from Karr et al. 1986).

Integrity class	Attributes	Total IBI score
Excellent	Comparable to the best habitats without human disturbance; all species expected for the region, stream size, and habitat, including the most intolerant forms, are present with a full array of age- (size-) classes; balanced trophic structure and few hybrids or diseased fish; very few or no introduced species	58–60
Good	Species richness somewhat below expectation, especially due to loss of the most intolerant forms; some species have less than optimal abundance or age and size distributions; trophic structure shows some signs of stress, but still few hybrids or diseased fish; the proportion of individuals that are members of introduced species is usually low	48–52
Fair	Signs of additional deterioration include loss of intolerant forms, fewer species, and generally reduced abundances; skewed trophic structure is indicated by increasing frequency of omnivores and tolerant species (e.g., green sunfish *Lepomis cyanellus*); older age-classes, especially of top carnivores, may be rare; incidence of hybrids and diseased fish increases above natural levels; the proportion of individuals that are members of introduced species increases	40–44
Poor	Community dominated by few species, many of which are omnivores, tolerant forms, habitat generalists, or introduced species; few top carnivores; many age-classes are missing, and abundance, growth, and condition are commonly depressed; incidence of hybrids and diseased fish is moderate	28–34
Very poor	Few fish present, most of which are introduced or tolerant omnivorous species; hybrids are common, and incidence of disease, parasites, fin damage, and other anomalies is high	12–22
No fish	No fish are captured, even after repeated sampling	

variation in the IBI were different in selected Illinois, Ohio, and West Virginia streams.

Examples of Use

After Karr's (1981) initial proposal of the IBI and the development of methods for applying it to other ecological regions (Fausch et al. 1984), investigators began to use it to assess various perturbations and to test it further in different regions. Karr et al. (1985) used the IBI to show that removal of residual chlorine (0.1–1.7 mg/L) from three Illinois sewage treatment plant effluents resulted in marked improvements in fish communities downstream. In contrast, they found that removal of moderate concentrations of unionized ammonia (0.01–0.18 mg NH_3-N/L) had little effect, and questioned the cost-effectiveness of tertiary treatment to remove ammonia in similar situations.

Leonard and Orth (1986) modified the IBI for use in tributaries of the New River, West Virginia, which support naturally depauperate fish communities due to high-gradient barriers and other zoogeographic factors. The seven streams they studied had a range of degradation due to sewage and mining pollution and sediment; the severity of degradation was independently assessed with an index of cultural pollution. They found that six original and modified metrics responded consistently over the range of degradation, and that a modified IBI developed from these attributes agreed closely with their cultural pollution index.

Berkman et al. (1986) used both macroinvertebrates and fish to assess stream quality in northeastern Missouri, where streams are degraded primarily by silt from intensive agriculture. Like Leonard and Orth (1986) they developed an independent index of habitat quality based on substrate, riparian vegetation, and land use. On the basis of detrended correspondence analysis and a diversity index, they found that benthic macroinvertebrates reflected habitat quality, but neither measure of community structure reflected habitat quality when fish were used. In contrast, the IBI was significantly correlated with their habitat quality index. They attributed the different results to a more direct effect of silt on macroinvertebrates than on fish and suggested that the IBI was more sensitive to indirect effects of silt on fish feeding and reproduction (Berkman and Rabeni 1987).

Steedman (1988) reported strong relationships among scores of a modified IBI and measures of urbanization, forest cover, and riparian forest for 10 watersheds near Toronto, Ontario. By analysis of data at several spatial scales, he found that land use in the 10–100 km^2 of watershed upstream from sites was most important in predicting site quality. He also used his models to address the landscape scale question of how much urbanization and riparian deforestation would be required to degrade sites beyond specific thresholds of biological integrity.

Several investigators have tested the IBI against known perturbations. Karr et al. (1987) found that the IBI detected effects from known channelization and wood debris removal, sewage effluents, and runoff of sediment, nutrients, and toxic chemicals from adjacent agricultural lands in three midwestern streams. In two of the streams, they found that the IBI ranked sites similarly during 3 years when habitat and water-quality conditions were relatively stable, based on samples from several seasons each year. Hughes and Gammon (1987) reported that a modified IBI reflected gradual degradation of substrate and water quality along the Willamette River, Oregon, more accurately than either species richness or Gammon's index of well-being. Hughes and Omernik (1981), Paragamian (1986), Nuhfer and Evans (1987), and Whittier et al. (1987) also found that the IBI reflected known regional patterns of stream degradation in the midwestern U.S.

Other investigators analyzed behavior of the IBI in different situations. Angermeier and Karr (1986) assessed the effects of sampling effort, and concluded that several pool–riffle sequences should be thoroughly sampled to reflect site quality accurately. Inadequate sampling resulted in underestimation of species richness because of patchy distributions of stream fishes among habitats (Angermeier 1987). In contrast, inclusion of age-0 fish inflated species richness, especially in years of high recruitment (Schlosser 1985, 1987) due to the greater probability of sampling rare species as young of the year (Angermeier and Karr 1986; Angermeier and Schlosser 1987). Angermeier and Schlosser (1987) and Karr et al. (1987) found that the IBI detected known degradation of water quality and habitat structure in two Illinois streams more consistently than the Shannon–Wiener diversity index. Angermeier and Schlosser (1987) also reported that omission of age-0 fish markedly affected assessments of site quality with the diversity index.

Advantages and Disadvantages

The IBI has several advantages, most of which have been presented in previous examples. First, the IBI is a broadly based ecological index that assesses both community structure and function at several trophic levels, and includes attributes of fish populations and the condition of individual fish. As a result, the IBI uses more of the information embodied in a community sample than do other indices. Second, the resulting IBI classes are biologically meaningful (Table 1), because sound biological judgment can be incorporated throughout the process of assessing biological integrity of a site (of course, a competent fish biologist must develop the IBI for the region and determine the scores and classes). Third, the IBI is flexible and has been applied to various ecological regions throughout the U.S. where stream fish communities are at least moderately diverse (Miller et al. 1988). Fourth, IBI metrics are sensitive to many types of degradation, including domestic and industrial sewage effluents; mining pollution; runoff of sediment, nutrients, and toxic chemicals from agricultural lands; and channelization, wood debris removal, and other habitat degradation. Finally, IBI scores are reproducible, having consistently ranked sites along known gradients of degradation similarly from year to year when no changes in habitat or water quality were evident (Karr et al. 1987). Moreover, Steedman (1988) reported low variation in site IBI scores calculated from samples within and between successive years.

There are also disadvantages of using the IBI. First, it requires complete and careful sampling of the fish community, so that all species are captured in proportion to their true relative abundance. Or, population estimates can be made for each species, as Leonard and Orth (1986) did. Sound sampling of fish communities is no trivial undertaking, but we repeat that it is also critical for applying all other indices. Second, fish communities must have at least moderate species richness; about five species at an undegraded site is a minimum. In communities naturally having fewer species, such as in many coldwater streams, attributes of population abundance and reproductive success and measures of stress or health for individual fish could be used. Third, developing the IBI for an ecological region re-

TABLE 2.—Nine primary underlying assumptions of the index of biotic integrity concerning how stream fish communities change with environmental degradation.

(1)	The number of all native species and of those in specific taxa or habitat guilds declines
(2)	The number of intolerant species declines
(3)	The proportion of individuals that are members of tolerant species increases
(4)	The proportion of trophic specialists such as insectivores and top carnivores declines
(5)	The proportion of trophic generalists, especially omnivores, increases
(6)	Fish abundance generally declines
(7)	The proportion of individuals in reproductive guilds requiring silt-free coarse spawning substrate declines, and the incidence of hybrids may increase[a]
(8)	The incidence of externally evident disease, parasites, and morphological anomalies increases
(9)	The proportion of individuals that are members of introduced species increases

[a] The incidence of hybrids was originally proposed by Karr (1981) to assess the loss of reproductive isolation due to degradation, but hybrids are difficult to identify and incidence may vary due to other factors. Recently, other investigators such as the Ohio EPA have followed Karr's (1981) suggestion of using metrics based on reproductive guilds to measure the loss of forms requiring clean spawning substrate.

quires background information on fishes and fish communities from a variety of streams, including relatively undegraded sites. However, the fish fauna in most regions of the USA and Canada has been described in regional ichthyological references (e.g., Baxter and Simon 1970; Scott and Crossman 1973; Eddy and Underhill 1974; Pflieger 1975; Moyle 1976; Smith 1979; Wydoski and Whitney 1979; Trautman 1981; Becker 1983; Smith 1985; Harlan et al. 1987); in all regions where we have worked, various agencies had collected fish community data (albeit often of varying quality) at different sites.

A fourth disadvantage often raised is that methods for establishing metric criteria, such as maximum species richness lines and intolerant species, are subjective. Several investigators have devised improved methods for setting some criteria. Notable are the efforts of Leonard and Orth (1986) to assess sensitivity of specific metrics to known degradation, and efforts of the Ohio EPA (OEPA 1988) to investigate behavior of all the IBI metrics as a function of stream size and sampling gear. The Ohio EPA also proposed objective methods for determining intolerant species, using samples from disturbed and least-disturbed sites throughout five Ohio ecological regions. This issue also points up the problem that because the IBI is a composite index, each of its components potentially suffers from the pitfalls previously described. For instance, intolerant species may be difficult to choose objectively and species richness depends on region, stream size, and sample size. Methods have been developed to obviate some of these problems, but others remain unsolved.

A fifth disadvantage is that the IBI must be modified for use outside the midwestern USA, where it was developed, and for aquatic environments other than lotic systems. We reiterate that recent modifications have largely retained the ecological framework of the original index (Miller et al. 1988). A sixth possible disadvantage is that the IBI is based on fish abundance and proportions thereof, but community structure and function may also be influenced by biomass, which has not been incorporated to date. Finally, the statistical properties of the IBI have not been well studied, although some research has indicated the range of natural variation to expect (e.g., Karr et al. 1987). For example, when the number of individuals captured at a site is low (e.g., <50), addition of a few fish may change some proportions (such as those of top carnivores or disease) enough to alter metric scores because the class intervals for scoring metrics are narrow (i.e., 0–1% top carnivores receives a score of 1). This renders the IBI score insensitive to small samples. If low sample size is due to degradation, other metrics should cause a low overall score anyway. If, however, low numbers are due to inadequate sampling, assessment with the IBI should await better data.

Recent Developments

When the IBI is reduced to its simplest concepts, it is based on nine major assumptions about how fish communities respond to degradation (Table 2). Further IBI testing should focus on the validity of these assumptions in different settings. A good example is the recent critical evaluation of parasitism by the digenetic trematode *Neascus* sp. ("blackspot disease") over a range of degradation. Leonard and Orth (1986), Nuhfer and Evans (1987), and Ohio EPA biologists (OEPA 1988) all found that incidence of this parasite, and perhaps others, bears little relation to degradation and instead appears to reflect habitat quality for the snails and fish that are intermediate hosts (Berra and Au 1978). These workers, therefore, deleted the incidence of *Neascus* when calculating the percentage of individuals with disease, parasites, and anomalies (OEPA 1988).

In like fashion, modification of the IBI for different habitats (e.g., lentic systems and estuaries: Greenfield and Rogner 1984; Thompson and Fitzhugh 1986) or different taxa (e.g., macroinvertebrate communities: OEPA 1988) would profit from development of a similar model of how the respective communities change with degradation. For instance, predictable changes in Great Lakes fish communities after degradation (Ryder and Edwards 1985; Marshall et al. 1987) might be incorporated into a method for assessing ecosystem change, as suggested previously.

Water resource management agencies in many U.S. states are evaluating the IBI for measuring biological integrity. As an example, the Ohio EPA has developed the IBI for monitoring water resources of that state (Whittier et al. 1987; OEPA 1988) and has designed a similar ecological index based on aquatic microinvertebrates. These efforts are a model for use of the IBI in practical water quality monitoring. Moreover, this agency has also proposed use of this index to set regulatory standards of stream quality (E. T. Rankin, Ohio EPA, personal communication), which would be the first such use.

Choosing the Best Method

A decision about the best method or index for assessing environmental degradation by means of data on fish communities depends on the goals of the study and the type and quality of data available. As for all investigations, it is essential that goals and required data are carefully considered before the study begins, that pilot data are evaluated to ensure that the entire fish community is representatively sampled (Green 1979; Hocutt 1978, 1981), and that sampling is carefully conducted to ensure accurate measurement of the fish community. No method can offset biases due to poor sampling.

Another requisite for applying any of the methods is inclusion of baseline data from relatively undisturbed fish communities, against which data from assemblages in degraded environments can be compared. These data are used to set expected values for other sites (as in determinations of indicator taxa, species richness, and IBI metrics) or become an integral part of the calculation of the index (as in multivariate analyses). Historical data collected by early workers are invaluable in this regard, although experienced ichthyologists must often be consulted to decipher outdated taxonomic nomenclature. In any event, the biologist cannot evaluate biological integrity effectively by any method without first addressing the question, "What *should* fish communities look like in this ecoregion?"

Given these prerequisites, when only qualitative data on presence or absence of fish species at specific sites are available, determinations of indicator taxa, species richness, or multivariate techniques for presence–absence data may be used to assess biological integrity (Table 3). When accurate quantitative data on relative abundances are available, but there is little background information about the fish community, one can use appropriate multivariate methods. We caution that these latter assessments depend on the unperturbed fish communities used for comparisons because those samples define the boundaries of multivariate space. However, ecological information and historical data on fish communities are available for most locations in North America and Europe. In contrast, the taxonomy, ecology, and distribution of fishes is poorly known in tropical regions. Where fish faunas are relatively diverse and well documented, we recommend using the IBI to assess environmental degradation (Table 3). Multivariate statistics can also be useful if appropriate samples from degraded and undisturbed fish communities are available or can be constructed, but little work has been done in this area to date. We do not recommend using species diversity or evenness indices because of their theoretical and statistical flaws and because they incorporate little biological information and lack ecological relevance. Instead, we suggest one of the other methods (Table 3).

Finally, whatever the method used to evaluate biological integrity, it is essential to incorporate the knowledge and experience of a competent fish biologist. Sound professional judgment prevents what Behnke (1987) termed the "illusion of technique," whereby results are accepted as fact because they were derived from a sophisticated algorithm even though they have little biological basis.

Future Research

We propose three main areas as having the highest priority for further research. First more effort must be devoted to standardizing methods of sampling fish communities in different environments (Hocutt et al. 1974; Hocutt 1978) and of data collection and analysis for several of the indices. The Ohio EPA (OEPA 1988) has made great strides in all these areas for the IBI.

Second, more research must focus on natural variation in fish communities over a range of

TABLE 3.—Recommended methods for use of fish communities to assess environmental degradation given different types of data and background biological information. All methods require accurate sampling of fish communities. Advantages and disadvantages of each method are discussed in the text. Species diversity and evenness indices are not recommended for this purpose for reasons given in the text.

Best method or index	Advantages	Disadvantages
	Qualitative data: presence or absence of fish species	
Indicator taxa or guilds	Conceptually simple and easily applied; guilds allow resolution of specific stresses	Few guidelines for choosing taxa; taxa may be absent for reasons other than degradation; sensitivity of taxa may vary by region or season; sensitivity is relative to other species present; taxa may be sensitive only to certain stresses; indicator taxa cannot be used to assess relative degradation; conveys little information
Species richness	Conceptually simple and easily applied	Dependent on sample size; conveys little information when used alone; varies by region
Multivariate procedures	Quantitative sophistication; simultaneously compares all samples	May be complex and difficult to interpret; few standard procedures, and statistical properties of some methods poorly known; results depend on reference samples used
	Quantitative data: relative abundance, but little biological information	
Multivariate procedures	Can compare community structure, or function, or both; can use abundance or biomass; all the advantages of multivariate procedures above	All the disadvantages of multivariate procedures above
	Quantitative data: relative abundance, with pertinent biological information	
Index of biotic integrity	Broadly based ecological index that assesses structure and function; integrity classes are biologically meaningful; flexible and widely applicable; metrics sensitive to different sources of degradation; scores are reproducible	Requires at least moderate species richness; requires background ecological information; methods for setting some criteria are subjective; must be modified for each ecological region; measures of biomass are not included; statistical properties are not well studied

degradation so that biologists can more accurately assess the significance of changes caused by perturbations. As in statistical analysis, one can only determine the significance of variation among treatments by comparisons to natural and sampling variation within treatments.

Third, as with virtually all problems in ecology, experimental manipulations with proper controls and replicates are needed to test assumptions underpinning all the indices (e.g., Table 2). Because large-scale manipulations are often difficult, unethical, or impossible to carry out, investigators should take advantage of ongoing modifications of land and water resources, such as channelization and wood debris removal (Karr et al. 1987), addition or removal of sewage effluents (Karr et al. 1985), dams, and flow fluctuations. As Karr et al. (1987) stated, these cultural perturbations provide experimental manipulations that no biologist can duplicate. But these manipulations often fall short of premeditated experimental designs due to lack of pretreatment data and randomized replication.

The most practical and achievable experimental manipulations are those that improve habitat or water quality and are expected to improve biotic integrity of aquatic communities. For example, we suggest that researchers measure changes in fish communities (referenced to control sites) in situations where effluents are to be upgraded (Karr et al. 1985) and where agricultural land is to be removed from intensive tillage. Relatively simple habitat manipulations such as those reported by Angermeier and Karr (1984) also hold promise for elucidating fish community response.

Evaluations of the methods we present have not yet reached these goals. For example, the best tests of assumptions underpinning the IBI have been of several types. Several investigators analyzed attributes of relatively undisturbed stream fish communities in Illinois and Kentucky (Fausch et al. 1984) and Ohio (OEPA 1988) to set expected values for metrics. Other researchers assessed changes in fish communities along known gradients of perturbation (Leonard and Orth 1986) or at sites with known drastic degradation (Karr et al. 1987). Yet, we clearly have much to learn.

Given the interest of water-resource managers in determining environmental quality by direct measurement of aquatic biota, we feel confident that researchers will continue to develop and improve ecological methods that use fish and other organisms to assess biological integrity. In our view, the goal of these efforts should be to develop viable tools that allow biologists to provide significant input into decisions about water-resource management.

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Appendix: Community Indices

TABLE A.1.—The most commonly used indices of species diversity and evenness and of community similarity.

Index name	Equation[a]	Reference
	Species diversity and evenness	
Shannon-Wiener	$H' = -\sum_{i=1}^{s} \frac{n_i}{n} \log_e \frac{n_i}{n}$	Shannon and Weaver (1949)[b]
Brillouin	$H = \frac{1}{N} \log_e \left(\frac{N!}{N_1! \cdot N_2! \cdot \ldots \cdot N_i!} \right)$	Brillouin (1951)[b]
Evenness[c]	$J' = \frac{H'}{H'_{max}} = \frac{H'}{\log_e s}$	Pielou (1975)
Diversity of change	$H\Delta' = -\sum_{i=1}^{s} \frac{\lvert\Delta n_i\rvert}{\sum^{s} \lvert\Delta n_i\rvert} \cdot \log_e \left[\frac{\lvert\Delta n_i\rvert}{\sum^{s} \lvert\Delta n_i\rvert} \right]$	Cornell et al. (1976)
Evenness of change	$J\Delta' = \frac{H\Delta'}{H\Delta'_{max}} = \frac{H\Delta'}{\log_e s}$	Cornell et al. (1976)
Index of well-being	$\text{Iwb} = 0.5 \log_e n + 0.5 \log_e w + H'_n + H'_w$	Gammon (1976); Gammon et al. (1981)
	Community similarity	
Jaccard similarity coefficient	$\text{JSC} = \frac{a}{a+b+c}$	Jaccard (1908)[b]
Percentage similarity coefficient	$\text{PSC} = 100(1.0 - 0.5 \sum_{i=1}^{s} \lvert p_{ix} - p_{iy}\rvert)$ $= 100 \sum_{i=1}^{s} \min(p_{ix}, p_{iy})$	Whittaker (1952)

[a]Variables in equations are:
- s = number of species in a sample from the community;
- n = number of individuals in a sample from the community;
- n_i = number of individuals of species i in a sample from the community;
- N = total number of individuals in the community;
- N_i = number of individuals of species i in the community;
- $\Delta n_i = n_{i(t+1)} - n_{i(t)}$ = difference in relative abundance of species i in two successive samples at times t and $t+1$;
- w = total biomass of fish;
- H'_n = Shannon-Wiener diversity based on fish numbers;
- H'_w = Shannon-Wiener diversity based on fish biomass;
- a = number of species found in both sample x and sample y;
- b = number of species found in sample x but not sample y;
- c = number of species found in sample y but not sample x;
- p_{ix} = proportion of species i in sample x;
- p_{iy} = proportion of species i in sample y.

[b]Cited in Washington (1984).

[c]Evenness shown only for Shannon-Wiener diversity. See Pielou (1975) for an evenness equation that applies to the Brillouin diversity index.

TABLE A.2.—Metrics of the original index of biotic integrity used to assess stream fish communities in the midwestern USA. (After Karr 1981; Fausch et al. 1984; Karr et al. 1986.)

Category	Metric	Scoring criteria[a] 5	3	1
Species richness and composition	(1) Total number of fish species	Expectations for metrics 1–5 vary with stream size and region.		
	(2) Number and identity of darter species			
	(3) Number and indentity of sucker species			
	(4) Number and indentity of sunfish species			
	(5) Number and indentity of intolerant species			
	(6) Proportion of individuals that are green sunfish	<5%	5–20%	>20%
Trophic composition	(7) Proportion of individuals that are omnivores[b]	<20%	20–45%	>45%
	(8) Proportion of individuals that are insectivorous cyprinids	>45%	20–45%	<20%
	(9) Proportion of individuals that are piscivores (top carnivores)	>5%	1–5%	<1%
Fish abundance and condition	(10) Number of individuals in sample	Expectations for metric 10 vary with stream size and sampling methods[c]		
	(11) Proportion of individuals that are hybrids	0%	>0–1%	>1%
	(12) Proportion of individuals with externally evident disease, parasites, or other anomalies	<2%	2–5%	>5%

[a]Ratings of 5, 3, or 1 are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the value expected at an undisturbed site in a stream of similar size in the same ecological region.

[b]Omnivores are defined as species with diets composed of at least 25% plant and at least 25% animal material (Karr et al. 1986).

[c]See Fausch et al. (1984) and Karr et al. (1986) for guidance in setting criteria for this metric.